The Effects of Motorized Watercraft on Aquatic Ecosystems

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Introduction

What do we mean by "motorized watercraft?"

Motorized watercraft include powerboats, fishing boats, pontoon boats, and "jet skis" or personal watercraft (PWC). They are propelled by some sort of motor: outboard, inboard, inboard/outboard, or jet propulsion. Most of these propulsion systems make use of a propeller. In the discussion of impacts presented here, all craft will be lumped together as "boats," unless otherwise stated (for example, see special section on PWCs). "Boat activity" refers to the ways in which these watercraft are used: fishing, cruising, water-skiing, racing. No distinction will be made between the types of activities unless otherwise stated.

Why are motorized watercraft important to aquatic ecosystems?

There are a number of reasons why boats and boat activity are an important issue. Numbers of registered boats in Wisconsin have increased by 87% since the late 1960's (567,000 in 1997-98 compared to 303,000 in 1968-69). Size of boats has also increased: over 40% of the registered boats were between 16 and 39 feet long in 1997-98 compared to just 18% in 1968-69. Along with the bigger boats have come bigger engines. The Duluth News-Tribune reports that horsepower has doubled on new boats registered in MN between 1981 and 1999. There has also been an explosion in recent years in new types of watercraft, especially personal watercraft. PWCs in WI increased from 6500 in 1991 to 28,900 in 1998, representing 5.1% of all registered watercraft. These smaller, more powerful craft have unique issues, due to their maneuverability and accessibility to shallow and remote areas. Finally, increased development of lakes and rivers leads to increased boat activity, especially in areas that have traditionally not been used for recreation.

How might boats affect aquatic ecosystems?

Boats may interact with the aquatic environment by a variety of mechanisms, including emissions and exhaust, propeller contact, turbulence from the propulsion system, waves produced by movement, noise, and movement itself. In turn, each of these impacting mechanisms may have multiple effects on the aquatic ecosystem. Sediment resuspension, water pollution, disturbance of fish and wildlife, destruction of aquatic plants, and shoreline erosion are the major areas of concern and will be addressed in the following pages. Impacts of boats that primarily affect human use of lakes, such as crowding, safety, air quality, and noise will not be addressed specifically.

As we discuss the impacts and effects of boats on the aquatic environment, we need to recognize that:
1) boating is a highly valued recreational activity in Wisconsin ($200 million spent on boating trips per year, $250 million on equipment);
2) most people use boats for fishing (58%);
3) public access is important and actively encouraged by the State of Wisconsin;
4) many of the issues associated with boating are complex, with sociological as well as ecological consequences; and
5) boating activities must be evaluated in the context of the characteristics of each waterbody and other factors that may be more important for the overall health of the aquatic ecosystem.

How is this document organized?

I have organized the material in this document in terms of the aspect of the aquatic ecosystem that may be affected by boat activity. The sections include:

A. **Water Clarity (Turbidity, nutrients, and algae)**
B. **Water Quality (Metals, hydrocarbons, and other pollutants)**
C. **Shoreline Erosion**
D. **Aquatic Macrophytes (Plant communities)**
E. **Fish**
F. **Aquatic Wildlife**
G. **Personal Watercraft ("Jet skis")**

Each section includes an introduction, a summary of three to five studies relevant to the issue, some conclusions, and a list of additional references for further reading. The introduction attempts to define the issue, explain why it is important to aquatic ecosystems and identify factors that affect it, and summarize some of the particular concerns related to boat activity. The conclusion summarizes the current state of knowledge, identifies uncertainties, and suggests management strategies that may be useful to deal with the issue. At the end of the document, I have included a summary section that incorporates information gleaned from all of the individual sections. A complete list of all studies mentioned in the text is given in the last section, entitled "For Further Reading."

A. Water Clarity (Turbidity, nutrients, and algae)

Introduction:

What do we mean by "water clarity?"
Water clarity is a measure of the amount of particles in the water, or the extent to which light can travel through the water. There are many ways to express water clarity, including Secchi disk depth, turbidity, color, suspended solids, or light extinction. Chlorophyll *a*, a pigment found in all plants, is often used to determine the amount of algal growth in the water and is related to water clarity as well.

Why is water clarity important in aquatic ecosystems?
Water clarity is important for a number of reasons. It affects the ability of fish to find food, the depth to which aquatic plants can grow, dissolved oxygen content, and water temperature. Water clarity is often used as a measure of trophic status, or an indicator of ecosystem health. Water clarity is important aesthetically and can affect property values and recreational use of a waterbody.

What factors affect water clarity?
Algal growth, runoff, shoreline erosion, wind mixing of the lake or river bottom, and tannic and humic acids from wetlands can all affect the clarity of the water. Water clarity often fluctuates seasonally and can be affected by storms, wind, normal cycles in food webs, and rough fish (e.g. carp, suckers, and bullheads).

How might boats affect water clarity?
Propellers may disturb the lake or river bottom directly, or indirectly through the wash or turbulence they produce, especially in shallow water. This may affect water clarity by increasing the amount of sediment particles in the water or may cause nutrients that are stored in the sediments, such as phosphorus, to become available for algal growth. Waves created by watercraft may contribute to shoreline erosion, which can cloud the water.

Studies:

Yousef and others (1980) is the most often cited publication on motor boat impacts. Turbidity, phosphorus, and chlorophyll *a* (chl *a*) were measured on control and intentionally mixed sites on three shallow Florida lakes (all less than 6 m or 18 ft deep), both before and after a set level of motor boat activity. On the two shallowest lakes, significant increases were seen in these parameters on the mixed sites, but not at the control sites. Average increases in phosphorus ranged from 28 to 55%. Maximum increases in turbidity and phosphorus occurred within the first two hours of boating activity. Turbidity declined at a slower rate after boating ceased, taking more than 24 hours to return to initial levels.

Hilton and Phillips (1982) developed an empirical model to predict the amount of turbidity generated by boats passing a stretch of river based upon field measurements of turbidity and timing of boat passes. The model assumes that each boat pass generates the same amount of turbidity and that it decays exponentially with time, such that the amount of turbidity at a given time is dependent upon the timing of the last boat pass. Using the model with maximum expected boat activity, the authors determined that turbidity returned to background levels 5.5 hours after cessation of boat movement, indicating long term build-up of turbidity was unlikely. The model also predicted that on an annual basis, 8 to 44% of the turbidity in the river could be attributed to motorboat activity, depending upon the amount of algal growth that occurred at the test sites.

Johnson (1994) investigated the role of recreational boat traffic in shoreline erosion and turbidity generation in the Mississippi River. Turbidity was monitored at several depths and distances from shore during weekends of heavy boating activity. Turbidity increased the most near the bottom of the river, but did not vary with distance from shore. Peak turbidity corresponded with peak boating activity, but only in sites with high boating activity.

U. S. Army Corps of Engineers (1994) investigated the relationship between boat traffic and sediment resuspension on the Fox River Chain O' Lakes in northeastern Illinois. Samples were collected in channels connecting the lakes so that boats could be counted with some accuracy. There was a direct correlation between the number of boat passes and the amount of suspended solids in the water column. However, the amount of resuspension varied with water depth and sediment type. In silt substrate, the highest amounts were seen in water depths of 3 ft, about half as much at 6 ft, and none at 8 ft. In marl substrate, effects were seen at 3 ft, but not 6 or 8 ft. The authors also determined that sediment resuspension by boats at 3 ft was equivalent to the amount of disturbance generated by a 20 mph wind, but that the frequency of boat passes was much higher than the frequency of winds of that magnitude.

Asplund (1996) investigated the effects of motor boats on sediment resuspension and concurrent effects on nutrient regeneration and algal stimulation in several Wisconsin lakes. Weekend and weekday water quality was measured on 10 lakes during three summer holiday weekends and an additional weekend in August. Motor boat use increased on holiday weekends compared to weekdays (200-350% increase). Water clarity usually decreased, associated with increases in turbidity, particularly in near-shore sites. Chl *a* showed no consistent trends. Phosphorus (TP) often increased in the mid-lake sites, while ammonia generally decreased in both areas. Shallower lakes tended to experience greater changes in turbidity and TP than deeper lakes. Water clarity and boat activity were measured on an additional 20 lakes during every summer weekend. Motor boat use increased consistently on weekends for most of the lakes in the study. Water clarity did not show a consistent increasing or decreasing trend for any individual lake on weekends. However, weekend Secchi disk readings were 10% lower than weekday readings on average for the entire data set. Clear water lakes tended to show slightly larger drops in clarity than turbid lakes, and had more weekends with decreased clarity. The magnitude of change in water clarity was small compared to seasonal changes and differences among lakes.

Conclusions:

What do we know?
Boats have been shown to affect water clarity and can be a source of nutrients and algal growth in aquatic ecosystems. Shallow lakes, shallow parts of lakes and rivers, and channels connecting lakes are the most susceptible to impacts. Depth of impact varies depending upon many factors including boat size, engine size, speed, and substrate type. Few impacts have been noted at depths greater than 10 feet.

What don't we know?
Less certain is the overall impact boats have on water clarity compared to other factors such as shoreline development, watershed runoff, storm events, and natural food web cycles. The cumulative impacts of boats on water clarity are also uncertain, as is the link between increased sediment resuspension and algal growth. Translating effects observed under experimental conditions to what happens under actual conditions can be difficult.

What can we do about it?
No-wake zones in shallow areas of lakes and rivers could help to reduce impacts on water clarity, both by reducing the overall amount of boat activity in these areas and by limiting impacts from high-speed boats. In certain cases it may be beneficial to restrict boat activity altogether, such as in extremely shallow waters where boats can disturb the bottom even at no-wake speeds.

Also see:

Garrad, P. N. and R. D. Hey. 1988. River management to reduce turbidity in navigable Broadland rivers. J. Environ. Manage. 27:273-288.

Gucinski, H. 1982. Sediment suspension and resuspension from small-craft induced turbulence. U.S. EPA Chesapeake Bay Program, Annapolis MD. 61 pp. (EPA 600/3-82-084)

Moss, B. 1977. Conservation problems in the Norfolk Broads and rivers of East Anglia, England - phytoplankton, boats, and the causes of turbidity. Biol. Conserv. 12:95-114.

B. Water Quality (Metals, hydrocarbons, and other pollutants)

Introduction:

What do we mean by "water quality?"
By water quality, we are referring to the chemical nature of a water body, particularly as affected by anthropogenic (human) sources. Metals (lead, cadmium, mercury), nutrients (phosphorus, nitrates), and hydrocarbons (methane, gasoline, oil-based products) can all be added directly to the water column through a number of sources, including boat motors. These added chemicals can affect other parameters, such as pH and dissolved oxygen.

Why is water quality important in aquatic ecosystems?
As discussed earlier, nutrients can affect the algal growth in lakes and rivers and have an effect on water clarity. Dissolved oxygen and pH levels influence the type and abundance of fish. In high enough amounts, metals and hydrocarbons can be toxic to fish, wildlife, and microscopic animals. In addition, these substances may have human health effects if a lake or reservoir is also used as a drinking water supply.

What factors affect water quality?
Runoff from watersheds, both urban and agricultural, is a major source of nutrients, pesticides, metals, and hydrocarbons in aquatic ecosystems. Point sources of pollution (from industrial or municipal wastes) are also common, especially in river systems. Even remote lakes can be affected by atmospheric deposition of metals and acid-producing chemicals.

How might boats affect water quality?
Boat engines are designed to deliver a large amount of power in a relatively small package. As a result, a certain amount of the fuel that enters into a motor is discharged unburned, and ends up in the water. Two-stroke engines, which make up a vast majority of the motors in use on all types of watercraft, have been particularly inefficient. Estimates vary as to how much fuel may pass into the water column (25-30% is a reasonable average) and depends upon factors such as engine speed, tuning, oil mix, and horsepower. Other concerns include lowered oxygen levels due to carbon monoxide inputs, and spills or leaks associated with the transfer and storage of gasoline near waterbodies.

Studies:

Schenk and others (1975) used small (0.5 to 4 acres), shallow (4 to 12 feet deep) ponds to investigate impacts of motors on water quality. They ran motors continuously for three years at a rate of 1 gallon of fuel per day per 1 million gallons of water (equivalent to 3 times the maximum likely boat activity on a heavily used lake). No changes were observed in standard water quality parameters (pH, nutrients), except due to scour of sediments, which caused elevations in alkalinity and hardness. Increased lead and hydrocarbon concentrations were detected in the water column and sediments of the test lakes. However, no acute toxicity was observed on any species. Phytoplankton growth, diversity, and species composition were unchanged. Zooplankton and bottom dwelling organisms were not affected. No changes in the fish community composition or mortality rates were exhibited.

Hallock and Falter (1987) measured nitrogen, carbon, and phosphorus levels in small enclosures after operating outboard engines in them for a period of time. Combining this information with estimates of the annual fuel consumption by motor boat users on a heavily used lake, they calculated the proportion of nutrient loading contributed by outboard motors. In this study, motorboat exhaust contributed about 1% of the total nitrogen loading to the lake, while the amount of phosphorus was negligible. On lakes which receive heavy use year-round (in the southern U.S.), motorboats could contribute up to 5% of the nitrogen loading. However, nutrient loading from other sources is much more significant.

Mastran and others (1994) determined the spatial distribution of polyaromatic hydrocarbons (PAH) in a reservoir used for both drinking water and recreation. Engine sizes are limited to a maximum of 10 horsepower in this reservoir. PAHs are a group of organic compounds found in petroleum products that can be released into the environment through combustion processes. Some of these PAHs are known to be carcinogenic, and thus of concern in a drinking water reservoir. The researchers found detectable levels of PAHs (up to 4 parts per billion) in the water column during times of peak boating activity (June), but not during October, when boat activity was minimal. PAHs were found in the sediments during both times, and tended to be higher in the vicinity of three marinas on the reservoir. Other sources of PAHs in the sediments could be from urban runoff and atmospheric deposition.

Reuter and others (1998) investigated the role of motorized watercraft on methyl *tert*-butyl ether (MTBE) levels in a California lake. MTBE is a fuel oxygenate required by many states to be added to gasoline to reduce carbon monoxide emissions in urban areas. MTBE is also a possible human carcinogen and imparts a noticeable taste and odor to drinking water in very low concentrations. The authors found that MTBE was detectable (0.1 μg/L) throughout the lake and throughout the year, but that it rose to 12 μg/L during mid-July in the upper waters of the lake, corresponding to peak boat use and the strongest stratification. This level exceeds drinking water standards under consideration in California. The authors determined that the exhaust from 2-stroke outboard motors was the primary source of MTBE, explaining 86% of the variability in MTBE levels. However, levels declined through the fall due to volatilization at the water surface and did not appear to persist from one year to the next.

Conclusions:

What do we know?

There have been numerous studies on the effects of outboard motor exhaust and related pollution from fuel leakage. (See **Wagner (1991)** for a good review of these studies.) In general, these studies have shown minimal toxic effects on aquatic organisms because 1) the amount of pollution is small compared to the volume of a lake; and 2) most hydrocarbons are volatile and quickly disperse. However, polyaromatic hydrocarbons and fuel additives have been detected in some cases, and could be a concern for drinking water supplies. Build-up of certain compounds in sediments has been documented, especially near marinas or other high concentrations of boats, and may be detrimental to bottom dwelling organisms.

What don't we know?

Most studies have focused on short-term or acute effects of outboard motor fuel and exhaust. Less clear are the long-term or chronic effects on organisms or human health of repeated exposure to low levels of pollutants.

What can we do about it?

Cleaner technology, such as four-stroke engines, and more efficient two-stroke models should help to reduce the inputs of fuel and exhaust into water bodies over time. Education of boaters and stricter controls of places that store and sell fuel near the water would help to reduce sediment contamination from fuel transfer and storage. Keeping engines well-tuned and using manufacturers' recommended mix of oil and gasoline would help engines run more efficiently and reduce the amount of unburned fuel that is discharged.

Also see:

Hilmer, T. and G. C. Bate. 1983. Observations on the effect of outboard motor fuel oil on phytoplankton cultures. Environmental Pollution 32:307-316.

Jackivicz, T. P. and L. N. Kuzminski. 1973. A review of outboard motor effects on the aquatic environment. J. Water Pollut. Control Fed. 45:1759-1770.

Wachs, B, H. Wagner, and P. van Donkelaar. 1992. Two-stroke engine lubricant emissions in a body of water subjected to intensive outboard motor operation. The Science of the Total Environment 116:59-81.

C. Shoreline Erosion

Introduction:

What do we mean by" shoreline erosion?"
Shoreline erosion is a term that refers to the process by which soil particles located along riverbanks or lakeshores become detached and transported by water currents or wave energy.

Why is shoreline erosion important in aquatic ecosystems?
Shoreline erosion may affect water clarity in near shore areas, shading submerged aquatic plants as well as providing nutrients for algal growth. It can interfere with fish use of shallow water habitat, as well as wildlife use of the land-water edge. Excessive shoreline erosion can negatively affect property values and can be expensive for riparian dwellers to prevent and control.

What factors affect shoreline erosion?
Shoreline erosion is affected by two main factors: 1) the intensity or energy of the erosive agent, i.e. water movement; and 2) the characteristics of the bank material itself. Water currents, waves, and water levels are the primary agents that cause shoreline erosion, although overland runoff can also erode shorelines. The erosivity characteristics of shoreline soils can also affect erosion rates – less cohesive materials such as sand erode more quickly than clay. The amount of vegetative cover, slope, and human disturbance also affect shoreline erosion rates at a given site. A certain amount of natural erosion may occur with storm or flood events, but usually erosion is minimal on natural shorelines. Shoreline development can affect erosion rates significantly by removal of vegetative cover or compaction of bank material.

How might boats affect shoreline erosion?
Boats produce a wake, which may in turn create waves that propagate outward until dissipated at the shoreline. Wave height and other wave characteristics vary with speed, type of watercraft, size of engine, hull displacement, and distance from shore. Propeller turbulence from boats operating in near shore areas may also erode shorelines by destabilizing the bottom.

Studies:

Bhowmik and others (1992) developed an equation to predict the maximum wave height of a recreational watercraft based upon the speed, draft, and length of the boat and the distance from a measuring point. Generally, the deeper the draft and longer the craft, the bigger the waves that were produced, while increased speed and distance diminished the size of the waves. During the controlled boat runs that were used to develop the model, wave heights averaged between 1 and 25 cm, with 10 to 20 waves produced per event. Maximum wave heights observed were up to 60 cm. During uncontrolled boating observations on the Mississippi and Illinois rivers, wave activity was observed to be continuous during peak boating times, with wave heights up to 52 cm.

Nanson and others (1994) monitored bank erosion and wave characteristics produced by three ferry boats in a set of staged boat passes to determine if speed limits on boat traffic could reduce river-bank erosion rates. Most of the measurements of the boat waves were positively correlated to rates of bank recession. Maximum wave height within a wave train was the simplest measure and was associated with a threshold in erosive energy at wave heights between 30 and 35 cm (12-14 in.). Above this threshold almost all bank sediments were observed to erode. Further monitoring revealed that reducing wave heights to < 30 cm, through speed limits on boats and reducing the frequency of boat passages, caused a decline in riverbank erosion. This threshold may vary from river to river depending upon the particle size and cohesiveness of the bank material.

Johnson (1994) placed iron stakes along transects in 1989 to monitor shoreline erosion along several stretches of the Mississippi River. Over a 3.5 year period, shoreline recession of up to 14 feet was observed in a channel subjected to intense boating activity (Main Channel) compared to less than 3 feet in a channel with similar river currents and light boating activity (Wisconsin Channel). [Author's update: Transects resurveyed in 1997 indicated 28 ft. of recession in the Main Channel compared to 4 ft. in the Wisconsin Channel. On average, the riverbank is eroding at a rate of 3 feet per year.]

Johnson and others (In preparation) investigated shoreline erosion due to recreational activity along several sites in the Lower St. Croix National Scenic Riverway. Over 4 successive boating seasons (1995-1998), 9 sites had net erosion, 2 sites had net deposition and 3 sites had no net change. When sorted by impact category, those sites with no boat waves and no foot-traffic trampling had sediment deposition or no net change in profile. Little net change was noted at sites with boat waves only. Shoreline erosion was documented at all sites with trampling only, as well as at all sites experiencing both waves and trampling. The surveys suggest that foot-traffic trampling and boat waves are major contributing influences to shoreline erosion in the study area. In the summer of 1998, additional investigations of off-peak and peak boating days included the measurement of maximum wave heights, number and type of boats, and shoreline sediment mobilization (erosion and resuspension). The study results confirmed that wave heights below 0.4 feet did not mobilize sediments, as determined in controlled run studies. However, the more boat waves 0.4 feet and higher in a 30 minute monitoring period, the greater the amount of sediment mobilized. Likewise, the larger the maximum wave height in a 30-minute monitoring period, the greater the amount of sediment mobilized. Of all the boat types recorded, runabouts and cruisers had the highest correlation to the measured maximum wave heights, amount of sediment mobilized, and number of waves greater than the sediment mobilization threshold (0.4 feet). Wind-generated waves above the threshold were not recorded during the study period.

Conclusions:

What do we know?
Waves or wake produced by boats is the primary factor by which boats can influence shoreline erosion. Wave heights depend upon speed, size and draft of boat, but can reach heights of 40-50 cm (15-20 in.) equivalent to storm-induced waves. However, wave heights dissipate rapidly as they move away from the boat, while wind waves increase with larger distances. Therefore, river systems, channels connecting lakes, and small lakes are likely to be most influenced by boat-induced waves, as boats may operate relatively close to shore and wind-induced waves are reduced. Shoreline erosion has been documented in river systems and has been attributed to frequency and proximity of boat traffic. Loosely consolidated, steep, unvegetated banks are more susceptible to shoreline erosion.

What don't we know?
It is unclear what effect boat waves have on shoreline erosion or bank recession in lake or still water environments. All studies to date have been on river systems. Also unknown is the cumulative impacts that boat waves can have on shorelines, especially in combination with wind-induced waves. While equations exist to predict how much of a wake a given boat can produce, very little information is available to suggest how much boat traffic a given shoreline can sustain. Also, individual boat waves may dissipate quickly, but boat traffic often mixes waves from several boats and can create much bigger waves that persist for longer periods of time.

What can we do about it?
No-wake zones are designed to minimize boat wake, so the obvious solution would be to use no-wake zones to limit shoreline erosion, particularly in channels or small sheltered lakes (i.e. areas where effective wind fetch is less than 1000 feet). Currently in WI, boats are restricted from operating at speeds greater than no-wake within 100 feet from fixed structures such as boat docks and swimming platforms. Many lake communities have established no-wake ordinances at 100 feet from shore or more. Seawalls and riprap have been used extensively in lakes and rivers to prevent shoreline erosion; however, these engineering approaches have little wildlife value and are expensive. Maintaining and restoring natural shorelines would help reduce the impacts of all types of waves on shoreline erosion.

Also see:

Bhowmik, N. G. 1976. Development of criteria for shore protection against wind-generated waves for lakes and ponds in Illinois. University of Illinois Water Resources Center Research Report No. 107, Urbana, IL. 44 pp.

Kimber, A., and J. W. Barko. 1994. A literature review of the effects of waves on aquatic plants. Natl. Biol. Surv., Environ. Manage. Tech. Center, Onalaska, WI. LTRMP 94-S002. 25 pp.

D. Aquatic Macrophytes (Plant communities)

Introduction:

What do we mean by "aquatic macrophytes?"
Aquatic macrophytes are large rooted plants that inhabit the littoral (shallow water) zone of most lakes and rivers. They are usually divided into three categories: submerged, emergent, and floating-leafed species. Common species include coontail, milfoil, elodea, pondweeds (submerged species), bulrushes, reeds, sedges, wild rice, and cattails (emergent), and water lilies, spatterdock, and lotus (floating).

Why are aquatic macrophytes important in aquatic ecosystems?
Aquatic plants perform many important ecosystem functions, including habitat for fish, wildlife, and invertebrates; stabilization of lake-bottom sediments and shorelines; cycling of nutrients; and food for many organisms. In some lakes, submerged plants grow in abundance, yet they also may compete with algae for nutrients and help maintain better water clarity. Emergent and floating-leafed species may be valued for their aesthetic qualities and help provide a more "natural" buffer between a developed shoreline and the open water.

What factors affect aquatic macrophytes?
There is considerable variability in plant communities, both within the same lake or river and among similar bodies of water. Macrophyte growth is limited by a number of factors, including light availability, nutrients, wave stress, bottom type, water level fluctuations, and water temperature. The shallow water extent of submerged plant growth is usually limited by bottom conditions and wave stress, while the deep water limit is usually dependent upon light availability. Eutrophication, boat traffic, controlled or raised water levels, shoreline development, invasive species, and rough fish can all have in impact upon aquatic plants, either through changes in abundance or species composition.

How might boats affect aquatic macrophytes?
Boats may impact macrophytes either directly, through contact with the propeller and boat hull, or indirectly through turbidity and wave damage. Propellers can chop off plant shoots and uproot whole plants if operated in shallow water. Increased turbidity from boat activity may limit the light available for plants and limit where plants can grow. Increased waves may limit growth of emergent species. Finally, boats may transport non-native species, such as Eurasian water milfoil, from one body of water to another.

Studies:

Zieman (1976) compared sea grass communities and sediment characteristics in undisturbed and motor boat disturbed areas off the Florida coast. Undisturbed sea grass beds had finer sediments than disturbed areas. In disturbed areas, channels receiving continuous boat traffic had coarser sediments than channels cut into the sea grass by a single boat pass. Sediments had lower pH and redox potential in the channels, indicating that removing aquatic vegetation altered sediment chemistry. As a result, channels cut by motor boats were found to persist for 2-3 years. Recolonization of disturbed areas was slow because of slow rhizome growth. Motor boat impacts are likely to be more pronounced in shallow high use areas with plant species that tend to be slow growing.

Murphy and Eaton (1983) looked at the relationship between boat traffic, turbidity, and macrophytes from several hundred sites in an English canal system. Abundance and biomass of macrophytes were negatively correlated to boat traffic, particularly at high levels (over 2000 boat passes per year). The impact on submerged vegetation was greater than on emergent plants. Total suspended solids were strongly correlated to boat traffic and negatively correlated to submerged macrophyte abundance, suggesting that boat traffic was indirectly suppressing macrophyte growth by generating turbidity. Direct physical damage by boats likely caused the decline in emergent macrophytes.

Vermaat and de Bruyne (1993) investigated factors that limited the distribution of submerged plants along three stretches of a lowland river in the Netherlands. Low light caused by high turbidity and periphyton growth, limited plants to water less than 1m deep. However, plant growth was much higher in the section that received the least amount of boat traffic, even though light conditions were similar to the other sites. In an experiment, plants collected from all three sites grew better in sheltered conditions than plants exposed to waves. The authors speculated that waves from boat traffic limited the shoreward extent of plant growth.

Mumma and others (1996) found a direct correlation between recreational use and drifting plants along stretches of the Rainbow River in Florida. Recreational use included canoeing, inner tubing, and motor boating, but no distinction was made among uses and their effect on the plants. Plants appeared to be damaged either by cutting or uprooting. However, the amount of plant biomass removed by the recreators per hour during peak use times represented a minute percentage of the total plant biomass in the upstream reaches of the river. Also, the researchers found that water depth and substrate type, not the level of use, influenced overall plant biomass among different sites.

Asplund and Cook (1997) studied the effects of motor boats on submerged aquatic macrophytes in Lake Ripley, Jefferson County, WI. Four enclosures, two of solid plastic and two of mesh fencing, were placed in about 1 m of water adjacent to high boat traffic areas. These enclosures were intended to exclude motor boat access and, in the solid-walled enclosures, to block the turbidity generated by boat-induced sediment resuspension. At the end of the study, plant biomass, height and percent cover were measured inside the enclosures and in control plots. Excluding motor boats from the experimental plots significantly increased macrophyte biomass, coverage, and shoot height compared to impacted areas. Results indicated that motor boats affected plant growth through scouring of the sediment and direct cutting; however, turbidity generated by boats did not appear to limit macrophyte growth in this experiment.

Conclusions:

What do we know?
Several researchers have documented a negative relationship between boat traffic and submerged aquatic plant biomass in a variety of situations. The primary mechanism appears to be direct cutting of plants, as many have noted floating plants in the water following heavy boat use. Other researchers have determined that scouring of the sediment, uprooting of plants, and increased wave activity may also be factors. Where frequent boat use has created channels or tracks, it was noted that these scoured areas persist for several years.

What don't we know?
While boats can uproot plants and reduce growth, it is still unclear what the long-term effects of boat traffic are on the macrophyte community, especially in lakes. Most studies that noted decreased plant growth in high boat traffic areas were in rivers where boat traffic is more confined and waves may be more of a factor. Also unknown is the effect on macrophyte species composition and the subsequent effect on other components of the aquatic ecosystem, such as the fish community and water quality. As one study noted, the amount of plant material chopped up by boats was a very small proportion of the whole plant community. It is unclear if such a small amount of plant material lost has larger-scale or longer-term impacts.

What can we do about it?
No-wake zones and restricted motor areas effectively reduce the impact of boats on aquatic plants (see **Asplund and Cook 1999**). Limiting boat traffic in areas with sensitive species or where a large proportion of the plant material is floating or emergent may be a good way to guide boat activity to more appropriate parts of a waterbody. While no-wake zones do not prevent all impacts, they do serve to reduce the overall amount of boat activity in a given area. Basing no-wake zones on water depth or the maximum depth of plant growth may be more useful than those based upon fixed distances from shore.

Also see:

Johnstone, I. M., B. T. Coffey, and C. Howard-Williams. 1985. The role of recreational boat traffic in interlake dispersal of macrophytes: A New Zealand case study. J. Environ. Manage. 20:263-279.

Schloesser, D. A., and B. A. Manny. 1989. Potential effects of shipping on submersed macrophytes in the St. Clair and Detroit Rivers of the Great Lakes. Mich. Academician 21:110-118.

E. Fish

Introduction:

What do we mean by "fish?"
In this discussion of boat impacts on fish or fish communities, we will consider impacts on a variety of levels: 1) individual fish, 2) fish populations, and 3) the community of all fish in a body of water. Aspects such as mortality and behavior affect individual fish, breeding success or recruitment affects fish population dynamics, and species composition and overall abundance of fish affect the fish community.

Why are fish important in aquatic ecosystems?
Fish form an important part of the food web in aquatic ecosystem, and can be either top predators, intermediate herbivores, or plankton eaters. A variety of birds and other animals depend upon fish as their primary food source. The presence or absence of individual species, as well as overall fish numbers can be an indicator of ecosystem health and can affect water clarity and water quality. Fisheries form an important resource for food and recreation for humans as well. In fact, angling is the most popular recreational activity on most Wisconsin waters.

What factors affect fish?
Climate, food availability and quality, suitability of shelter, and the presence of predators (including humans) affect individual fish, as well as fish populations. Water quality, turbidity, and the presence of pollutants can also affect fish reproductive success, which affects fish populations. Species composition is usually determined by a number of factors including water quality, water temperature, and pH. Angling also has a large impact on fish populations and community structure and is usually closely regulated to try to maintain a balanced fishery. In sum, any human activity that affects water quality and habitat has the potential to affect fish populations and overall community structure.

How might boats affect fish?
Direct contact of boats or propellers may be a source of mortality for certain fish species, such as carp. Pollution from exhaust or spills may be toxic to some fish species. Boat movement can affect individual fish directly by disturbing normal activities such as nesting, spawning, or feeding. Increased turbidity from boats may interfere with sight-based feeding or success of eggs or fish spawning. On a population level, boats may affect fish through habitat alteration caused by waves or propeller damage.

Studies:

Lagler and others (1950) addressed several important topics using control and experimental ponds: bluegill and largemouth bass production, location of nests, guarding behavior, mortality of eggs and fry, and habitat alteration. Some differences among motor and non-motor ponds were seen in fish production, but these differences were small and may have been due to other factors. The motor boat followed a defined path around the perimeter of the pond and thus inhibited macrophyte growth, scoured the sediments, and reduced the number of bottom dwelling organisms in its path. Otherwise, the motorboat ponds exhibited no changes in turbidity, water chemistry or phytoplankton production. Motorboat use did cause male sunfish to abandon their nests temporarily, but it did not affect the location of nests. Motorboat use did not significantly affect mortality of eggs or fry. Angling success was monitored on a non-motor lake on which a motor boat was operated every other day during several 3-week periods. No differences in angling success (either catch or strike frequency) were observed on motor vs. non-motor days.

Mueller (1980) used an underwater camera to record guarding behavior by sunfish in response to passes by a canoe, slow motorboat (2 mph), and fast motorboat (11 mph) at varying distances from nests. Boat passage caused fish to leave nests to take cover, leaving eggs vulnerable to predation. In control areas, fish left the nests just as often but for shorter periods of time, primarily to ward off intruders. Absence times were longer if boat passes were close or cover was far away. Fish abandoned nests more frequently in response to slower moving boats, most likely because of increased time for detection.

Kempinger and others (1998) studied the frequent occurrence of fish kills on a stretch of the Fox River in Oshkosh, WI, between Lake Butte des Morts and Lake Winnebago since the 1950's. Throughout the ice-free season in 1988, they monitored cages with fathead minnows and freshwater drum placed at various sites along the river. They discovered that an outboard-motor testing facility located along the river was primarily responsible for the fish kills, due to elevated levels of carbon monoxide in the water. Fish kills were most apparent during warm temperatures and low flow or reversed flow conditions due to incoming seiches from Lake Winnebago. As a result of the study, the testing facility now limits its testing to no more than 1500 horsepower at one time, and ceases operation during low flow and higher temperatures.

Conclusions:

What do we know?
Very few studies have documented direct impacts of boat activity upon individual fish behavior or mortality. The few studies cited here demonstrate that boat activity can disturb fish from their nests, but that overall breeding success is likely not affected. Toxic effects on fish have generally not been observed, except in extreme situations (such as near boat testing facilities). Of much greater concern and effort, however, is the effect of boats on fish habitat (water quality, clarity, and aquatic plants) which subsequently may impact fish populations. These studies have been summarized elsewhere.

What don't we know?
While the effects of boats on fish habitat has been studied extensively, as well as the effects of habitat degradation on fish populations, the link between boat activity and fish populations has not been well defined. How much boat activity can a lake or river handle before fish populations are affected? How much habitat is needed for successful fish recruitment? Is fishing success affected by boat activity? Would restricting boat activity enhance fish populations? These are questions that have not been addressed or answered to date.

What can we do about it?
Keeping boats out of known fish spawning areas may help to improve overall fish success, however, it would be detrimental to anglers. Most boat activity usually occurs after peak fish spawning times, but extending protection of critical areas through early June may help to protect certain species. A more useful approach would be to protect shallow waters and plant beds from boat activity through the use of no-wake zones. No-wake zones in prime fishing areas may also help to reduce user conflicts by creating a separation between anglers and high-speed boaters.

Also see:

Savino, J. F., M. A. Blouin, B. M. Davis, P. L. Hudson, T. N. Todd, and G. W. Fleischer. 1994. Effects of pulsed turbidity on lake herring eggs and larvae. J. Great Lakes Res. 20(2):366-376.

F. Aquatic Wildlife

Introduction:

What do we mean by "aquatic wildlife?"
Aquatic wildlife refers to animals that spend part or all of their life in aquatic environments, or depend upon them for food or reproduction. Examples include waterfowl, shorebirds, herons, eagles, loons, turtles, frogs, and in saltwater systems include manatees, seals, and dolphins. Fish will be addressed in a separate section.

Why are aquatic wildlife important in aquatic ecosystems?
Aside from the aesthetic value of being able to see eagles, loons, deer, and other animals near water, certain species form an essential part of the food chain, especially those that feed on detritus or carrion or those that feed on the top predator fish. The presence of loons and osprey can be an important indicator of ecosystem health.

What factors affect aquatic wildlife?
Wildlife use of aquatic ecosystems depends upon a number of factors. Good water quality and the availability of suitable habitat are important for most species. Other species require a certain amount of wild or natural area in order to find enough food or to be protected from predators. The quantity and quality of food is also essential. For example, loons need an abundant fish population in order to sustain their growth. Species that migrate may need a high quality food source in order to build up enough energy to reach their wintering grounds. Finally, some species are very sensitive to human presence and may not be able to survive on waters that are too "busy" or populated.

How might boats affect aquatic wildlife?
Boats may have direct impacts on wildlife through contact with propellers or disturbance of nests along the shoreline by excessive wave action. Disturbance by the fast movement of watercraft or even the presence of humans near feeding ground or breeding areas may prevent certain species, especially birds from being successful. Noise or harassment may cause some wildlife to vacate nests, leaving eggs or young vulnerable to predators. Indirect effects may include destruction of habitat or food source in littoral areas, or impaired water quality.

Studies:

Kahl (1991) describes detailed observations of the response of canvasbacks to fishing and hunting boats at feeding areas. Disturbances caused the flock to flush and reduced the amount of time the birds spent at feeding areas, possibly increasing energy costs and delaying migration. High frequency of disturbance caused the birds to establish refuge areas in the middle of the lake where they remained for up to 60 min. per disturbance. Boating disturbance accounted for ~50% of daylight hours spent away from feeding areas. Canvasbacks were less likely to flush and flushed at closer distances in response to slower moving boats.

Rodgers and Smith (1995, 1998) directly measured the flushing response of 16 waterbird species exposed to 5 different human activities, including walking, ATV, motorboat, canoe, and automobile. The earlier study focused on nesting birds, while the latter focused on foraging and loafing birds. The authors found considerable variation in flushing distances among different species in response to the same activity (mean distances ranging from 5 to 35 m). In general, birds which were more habituated to human presence (gulls, terns) exhibited the least flushing distance. Walking and canoeing tended to flush birds at greater distances than motorized activity, perhaps due to the slower speeds and more time for birds to become aware of the disturbance. Nesting birds tended to allow closer approaches before flushing, likely because of the greater cost of leaving a nest versus a feeding area. In both studies, the authors recommend buffer zones of 100 m to protect most bird species, or mixed colonies of either nesting or foraging birds. This figure includes a 40 m "buffer" to account for alarm behaviors that do not result in an actual flush.

Madsen (1998) studied the disturbance effects of a variety of recreational activities on coot, widgeon, and mute swan flocks in 2 Danish wetlands. Moving hunting boats caused the most disturbance in terms of flushing frequency (2 times per day on average) and disruption time (up to 75 minutes), compared to stationary boats, fishing, windsurfing, and sailing. However, windsurfing had the highest flushing distance of any activity (450-700 m). Widgeon and mute swan were disturbed much more easily than coots. Repeated disturbances during a day reduced foraging time by 13-33%. In terms of overall effects of recreational activity, birds were disturbed 16% of the daylight hours during the months of September and October.

Stalmaster and Kaiser (1998) observed the effects of recreational activity on wintering bald eagles in a wildlife area in northwest Washington. They observed fewer eagles and less feeding activity during times of highest recreational use (weekends, early morning hours). Foot traffic disturbed individual eagles to a greater extent than motor boats (greater flushing responses and distances), but boat activity disturbed a greater proportion of the eagle population. Eagles resumed feeding relatively quickly after initial disturbances of the day, but were slow to resume after about 20 disturbances. Boat activity was more disturbing on narrow than on wide river channels. The authors estimate that feeding by eagles was reduced by 35% in the wildlife area because of recreational use and suggest limiting boat traffic within 400 m of eagles, especially during early morning hours.

Conclusions:

What do we know?

Boat activity certainly causes many wildlife species to be disturbed from a variety of activities. For some species, this may represent just a temporary disturbance, with little long-term effect. For other species, or in cases where unique habitats are disturbed by high frequency or intensity of boat use, boat activity can have effects on the entire population. Migratory birds may require more protection as their energy needs can easily be disrupted by excessive disturbance. Manatees have been observed with scars and lesions from contact with boat propellers, but few other species likely receive this direct sort of impact.

What don't we know?

Very little research has been done on small animals that use shorelines, such as turtles, frogs, shorebirds, and mammals. Long term effects on wildlife use of an aquatic ecosystem is also difficult to assess, as motor boat activity often goes along with increased development and impaired water quality. Many species may simply move elsewhere if a particular body of water becomes too busy.

What can we do about it?

Buffer zones have been suggested for a variety of bird species, ranging from 100 to 180 m. Protecting littoral zone habitat or known breeding areas with no-wake zones would help to provide this buffer, though it would not eliminate boat activity. Preventing access to undisturbed shorelines or areas may be warranted if it can be shown that these areas provide a unique resource to wildlife populations. Loon nesting sites, heron rookeries, "turtle beaches," and eagle wintering sites, would all be possible candidates for such a restriction. In some cases, all human activity, not just motor boat use, may need to be restricted in order to protect wildlife populations.

Also see:

Bratton, S. P. 1990. Boat disturbance of ciconiiformes in Georgia estuaries. Colon. Waterbirds; 13(2):124-128.

Mikola, J., M. Miettinen, E. Lehikoinen, and K. Lentilä. 1994. The effects of disturbance caused by boating on survival and behaviour of velvet scoter *Melanitta fusca* ducklings. Biol. Conserv. 67: 119-124.

York, D. 1994. Recreational-boating disturbances of natural communities and wildlife: An annotated bibliography. U.S. Dept. of Interior, National Biological Survey, Biological Report 22. 30 pp.

G. Personal Watercraft ("Jet skis")

Introduction:

What do we mean by "personal watercraft?"
Personal watercraft (PWCs), commonly referred to as "jet skis", include a variety of watercraft that are designed for use by one or two individuals (though newer models are being developed for 3 people). Riders either sit or stand, depending upon the design. Propulsion systems are generally quite different from traditional outboard motors, making use of a water pump rather than propellers to move the craft through the water. Steering is accomplished by ejecting the water at high force through a movable nozzle. PWCs are designed to be powerful and maneuverable and can operate in waters less than 12 inches deep.

Why are PWCs important in aquatic ecosystems?
Since the introduction of the first Jet Ski in 1973, PWC use has skyrocketed throughout the country, especially since the late 1980's. It is estimated that 200,000 PWCs are sold annually in the U.S., representing 30% of all new sales of watercraft. They still represent a small proportion of overall watercraft in use (about 1 million compared to 12 million outboards), but on certain lakes and rivers, they can achieve relatively high numbers. Along with the increase in numbers has come increasing conflicts with other users, as they tend to be more noticeable and create noise and perceptions of reduced safety and increased crowding.

How might PWCs affect aquatic ecosystems?
PWCs can have many of the same effects as described in other sections. However, because of their unique propulsion systems and use characteristics, this special section has been included to summarize studies that have addressed the impacts of PWCs specifically. For example, PWCs are often criticized for the noise that they produce, due to their frequent stops and starts and operation at full throttle. Most PWCs employ two-stroke technology for their engines, thus making them a concern for their air and water emissions of hydrocarbons and other pollutants. Because PWCs can be operated in shallow water, at high speeds, and in remote areas not usually frequented by boats, disturbance to wildlife may be more of a concern than other types of watercraft. Finally, while PWCs do not generally have propellers, the turbulence produced by the jet propulsion may still disturb plant growth and sediments, especially during acceleration or turns when the thrust may be oriented downward.

Studies:

Noise
Wagner (1994) described a study of PWC noise vs. outboard motor noise on a heavily used lake. The study showed that the actual noise level (in terms of decibels) is not much higher than most other types of

watercraft. The loudness decreased with distance from the watercraft, such that the sound level was within background levels at distances of 300 feet or more. However, the PWCs tended to have more variable sound levels and a higher pitch than most other types of watercraft. These frequent changes in pitch tend to make the noise more noticeable to human ears, and were usually the cause of complaints. Responding to these concerns, PWC manufacturers have introduced quieter technology in recent years.

Disturbance to wildlife
Burger (1998) compared the effects of motorboats and personal watercraft on flight behavior over a colony of common terns on an island in Barnegat Bay, New Jersey. The presence of any watercraft caused birds to fly over the colony. However, personal watercraft caused more birds to flush than did motorboats, particularly early in the nesting season (150-200 birds for PWCs compared to 20-30 for boats). Racing and fast-moving watercraft elicited a higher response than slow moving boats, as did boats that operated outside of the established channel. More birds flew in the air the closer the approach by a boat or PWC. The proximity of watercraft and either the fast movement or noise of those operating at high speeds were the most disturbing attributes, and tended to be those associated with PWCs. These disturbances may cause a drop in breeding success for some colonies of terns.

Emissions
The **California Air Resources Board (1998)** has argued that emissions from PWCs on a per machine basis are actually higher than that for a typical outboard motor, due to their larger horsepower, higher speed of operation, and sustained high speeds. Estimates of 2-3 gallons of unburned fuel per hour are typical. However, it has been estimated that all outboard motors discharge 25-30% of their fuel unburned, not just PWCs. The actual amount discharged is a function of speed, tuning, size of engine and other factors.

Physical impacts
The **Personal Watercraft Industry Association (1997)**, found that PWCs had no effects on water clarity and seagrass disturbance in a shallow estuary at depths of 21-36 inches when operated on plane (20-30 mph). Some resuspension of fine sediments was documented during tests with frequent stops, starts, and turns in a confined area, however. This study only considered effects of single Jet Ski runs, and did not address cumulative impacts of sustained Jet Ski use in shallow water.

Conclusions:

What we do we know?
Available research into the impacts of PWCs on lakes and other water bodies is relatively limited. In general, the issues that are raised in regard to PWC use apply to all motorized watercraft. There is some evidence that noise and emissions are perhaps a bigger concern than for other types of watercraft, largely due to the way in which the machines are operated (high speed, frequent stops, starts, and turns). One study also showed that PWCs present a larger threat nesting waterbirds. PWCs may be more disturbing due to their ability to access areas typically avoided or restricted to other types of watercraft.

What don't we know?
Very few studies have been done which have documented physical impacts of PWCs on aquatic vegetation or sediment resuspension. No studies have compared the effects of PWCs to those of outboard motors. While PWCs may not have as much impact as a propeller-driven craft at a given depth, their operation in shallower water may have more overall effect. This area of concern remains to be addressed.

What can we do about it?
Manufacturers have voluntarily been introducing quieter, cleaner burning machines in response to citizen complaints and EPA rules requiring 75% reductions in air emissions from all marine engines by 2025. Wisconsin currently has a no-wake rule for PWCs within 200 feet of shore, which effectively minimizes the effect of PWCs on shallow water habitat. This no-wake restriction also reduces the noise level experienced by people on shore. Enforcement of this no-wake rule would go a long way toward minimizing the effects of PWCs. Restricting PWC use in natural areas or critical bird breeding areas may be justified in some cases; however restricting all motorized watercraft may be necessary to truly protect

species of concern. Some states and the National Park Service have considered or enacted bans on PWCs within their jurisdiction, largely based upon disturbance to wildlife and the noise issue.

Also see:

San Juan Planning Department. 1998. Personal Watercraft Use in the San Juan Islands. A Report Prepared for the Board of County Commissioners, San Juan County, Washington.

Summary Section

Potential mechanisms by which boats impact aquatic ecosystems and the effects that they can have on the aquatic environment. Shaded areas indicate where a "Mechanism" has an "Effect."

Mechanism: / *Effect:*	Emissions and exhaust	Propeller or hull contact	Turbulence	Waves and wake	Noise	Movement
Water Clarity (turbidity, nutrients, algae)						
Water Quality (metals, hydrocarbons, other pollutants)						
Shoreline Erosion						
Macrophytes (plant communities)						
Fish						
Wildlife (Birds, mammals, frogs, turtles)						
Human enjoyment (air quality, peace and quiet, safety, crowding)						

What do we know?

While the effects of boats on aquatic systems are complex and depend on a number of factors, a few general observations can be made. First, the physical effects of propeller, waves, and turbulence appear to be more of an issue than engine fuel discharge. Water clarity, aquatic plant disturbance, and shoreline erosion all are serious issues that can be exacerbated by boat traffic. Second, most of the impacts of boats are felt most directly in shallow waters (less than 10 feet deep) and along the shoreline of lakes and rivers not exposed to high winds (less than 1000 feet of open water). Third, these effects can have repercussions for other features of the aquatic ecosystem, including the fish community, wildlife use, and nutrient status. These observations all emphasize that the most important area of a lake or river to protect is the shallow-water, near-shore habitat known as the littoral zone. Boats that operate in deep waters with large surface areas are not likely to be impacting the aquatic ecosystem.

What don't we know?

Given these observations, there are still a number of unknowns regarding motor boat impacts. Most of the studies that are summarized here have focused on the short term or acute impacts of boat activity, pollution, disturbance, sediment resuspension, etc. It is not very clear what role boats can play in the long term changes of a water body, i.e. changes in macrophyte community, overall water quality, or fish and wildlife use. Many other factors influence these same features and many have changed along with boat activity. For example, increased shoreline development often causes increased boat activity, yet it is difficult to separate out which factor is more important for plant community changes. As another example, it has been demonstrated that boats and PWCs can disturb breeding bird activity, but it is difficult to determine what effect this may have on overall bird populations, due to the increasing amount of all human activities in historic breeding areas of many bird species.

What can we do about it?

While specifics of boat use management will be covered extensively in other chapters, we will make a few comments here regarding ways in which environmental impacts of boats can be reduced.

No-wake zones

Given that most impacts of boats are exhibited in shallow-water near-shore areas, protecting these areas with no-wake zones would be the most effective way of reducing impacts. No-wake zones have a dual benefit by both slowing boats down and directing traffic elsewhere. Currently in Wisconsin, boats are required to operate at no-wake speeds within 100 feet of piers, docks, and moored boats, while PWCs are required to operate at no-wake speeds within 200 feet of the shoreline. Lakes less than 50 acres in size are entirely no-wake. While established primarily for safety and navigation reasons, these restrictions appear to be adequate for protecting against shoreline erosion, at least in developed lakes. In many cases, however, these restrictions do not adequately protect shallow-water sediments or beds of aquatic macrophytes. Some communities have extended no-wake restrictions to 200 or even 300 feet through local ordinances. These extended no-wake areas have the potential to protect a much more significant proportion of the littoral zone and may help to reduce shoreline erosion.

A much more useful way of establishing a no-wake area would be to determine the depth at which plants grow in a given waterbody, and then establish a no-wake zone based upon water depth and vegetation parameters. At minimum, a no-wake zone based upon a 6-foot depth would reduce disturbance to sediments. A deeper depth threshold could be justified if the tops of plants come within 5 feet of the surface, or if the sediments were particularly fine. These guidelines could then be coupled with the minimum 100-foot no-wake zone to protect shorelines.

Restricted areas

In some cases, protection of aquatic resources may require restricting all boat activity, not just speed. Boats can still disturb plants, sediments, and wildlife at no-wake speeds. These types of restrictions need to be based upon unique features of a resource and are often used to provide a certain type of experience on remote or "wild" lakes. For example, to adequately protect waterbird breeding areas, a "buffer zone" of at least 100 m (300 feet) has been suggested, in which all human activity would be banned. Similar areas could be established for emergent or floating-leafed plant beds, which may be impacted by boats operating at any speed. Research on Long Lake in the Kettle Moraine State Forest – Northern Unit showed that no-motor zones did a better job of preventing disturbance of submerged plants than simple no-wake zones (Asplund and Cook 1999). Some lakes currently have electric-motor only or no-boat restrictions, which may help to protect particularly unique or sensitive natural areas. These types of restrictions need to balance protection of the resource with the right of public access.

Enforcement and Education

Many of the environmental problems associated with boat activity could be resolved with better enforcement of existing ordinances or regulations and promoting awareness among boaters. Slow-no-wake rules are often ignored or misunderstood by boaters, such that impacts to sediments, aquatic plants, and shorelines occur even in no-wake zones. Another important avenue is informing recreators about the value of plants, littoral zones, and natural shorelines and how their activities may affect the aquatic ecosystem. If people understand that their activities may be hurting the ecosystem, they may be willing to confine their activities to more appropriate places.

Technology

Recent technology spurred by Federal air quality standards has the potential to reduce water pollution impacts from outboard motors as well. All 2-stroke engine manufacturers, including traditional outboard motors and PWCs, must reduce air emissions by 75% by the year 2025. Most manufacturers have already introduced cleaner burning 2-stroke engines and PWCs. Four-stroke engines, which use fuel more efficiently, produce cleaner exhaust, and run more quietly than traditional 2-stroke engines, are becoming much more common. However, technology may have the opposite effect on physical impacts, as engine sizes continue to increase and PWC manufacturers continue to emphasize speed and power. The consequences of operating bigger and faster machines in our inland waterways must continually be addressed in the future.

For further reading:

Asplund, T. R. 1996. Impacts of motorized watercraft on water quality in Wisconsin lakes. Wis. Dep. Nat. Res. Bur. Research, Madison, WI. PUBL-RS-920-96. 46 pp.

Asplund, T. R., and C. M. Cook. 1997. Effects of motor boats on submerged aquatic macrophytes. Lake and Reserv. Manage. 13(1): 1-12.

Asplund, T. R., and C. M. Cook. 1999. Can no-wake zones effectively protect littoral zone habitat from boating disturbance? Lakeline, 19(1): 16-18+.

Bhowmik, N. G., T. W. Soong, W. F. Reichelt, and N. M. L. Seddik. 1992. Waves generated by recreational traffic on the Upper Mississippi River System. Report by the Illinois State Water Survey, Champaign, Illinois, for the U.S. Fish and Wildlife Service, Environmental Management Technical Center, Onalaska, WI. EMTC 92-S003. 68 pp.

Burger, J. 1998. Effects of motorboats and personal watercraft on flight behavior over a colony of common terns. The Condor, 100 (3): 528-534.

California Air Resources Board. 1998. Draft proposal summary, proposed regulations for gasoline spark-ignition marine engines 2 (June 11, 1998). Mobile Source Control Division, California Air Resources Board.

Hallock, D. and C. M. Falter. 1987. Powerboat engine discharges as a nutrient source in high-use lakes. USGS Res. Tech. Comp. Rep. 14-08-000-G1222-06. Washington, D.C. 45 pp.

Hilton, J. and G. L. Phillips. 1982. The effect of boat activity on turbidity in a shallow broadland river. J. Appl. Ecol. 19:143-150.

Johnson, S. 1994. Recreational boating impact investigations - Upper Mississippi River System, Pool 4, Red Wing, Minnesota. Report by the Minnesota Department of Natural Resources, Lake City, Minnesota, for the National Biological Survey, Environmental Management Technical Center, Onalaska, WI. EMTC 94-S004. 48 pp. + appendices.

Johnson, S., and others. In prep. St. Croix River shoreline studies, 1995 – 1998: Executive summary. Minnesota Department of Natural Resources.

Kahl, R. 1991. Boating disturbance of canvasbacks during migration at Lake Poygan, WI. Wildlife Society Bull. 19(3):242-248.

Kempinger, J. J., K. J. Otis, and J. R. Ball. 1998. Fish kills in the Fox River, Wisconsin, attributable to carbon monoxide from marine engines. Trans. Amer. Fish. Soc. 127:669-672.

Lagler, K. F., A. S. Hazzard, W. E. Hazen, and W. A. Tompkins. 1950. Outboard motors in relation to fish behavior, fish production, and angling success. Trans. N. Am. Wildl. Conf. 15:280-303.

Madsen, J. 1998. Experimental refuges for migratory waterfowl in Danish wetlands. I. Baseline assessment of the disturbance effects of recreational activities. J. Applied Ecology 35(3): 386-397.

Mastran, T. A., A. M. Dietrich, D. L. Gallagher and T. J. Grizzard. 1994. Distribution of polyaromatic hydrocarbons in the water column and sediments of a drinking water reservoir with respect to boating activity. Wat. Res. 28(11): 2353-2366.

Mueller, G. 1980. Effects of recreational river traffic on nest defense by longear sunfish. Trans. Amer. Fish. Soc. 109(2):248-251.

Mumma, M. T., C. E. Cichra, and J. T. Sowards. 1996. Effects of recreation on the submersed aquatic plant community of Rainbow River, Florida. J. Aquat. Plant Manage. 34:53-56.

Murphy, K. J. and J. W. Eaton. 1983. Effects of pleasure-boat traffic on macrophyte growth in canals. J. Appl. Ecol. 20: 713-729.

Nanson, G. C., A. von Krusenstierna, E. A. Bryant and M. R. Renilson. 1994. Experimental measurements of river-bank erosion caused by boat-generated waves on the Gordon River, Tasmania. Regulated Rivers: Res. and Manage. 9(1):1-15.

Personal Watercraft Industry Association. 1997. Effects of personal watercraft operation on shallow-water seagrass communities in the Florida Keys: Executive Summary. Prepared by Continental Shelf Associates, Inc. Jupiter, FL.

Reuter, J. E., and many others. 1998. Concentrations, sources, and fate of the gasoline oxygenate methyl *tert*-butyl ether (MTBE) in a multiple-use lake. Environ. Sci. Technol. 32:3666-3672.

Rodgers, J. A., and H. T. Smith. 1995. Set-back distances to protect nesting bird colonies from human disturbance in Florida. Conservation Biology, 9(1): 89-99.

Rodgers, J. A., and H. T. Smith. 1997. Buffer zone distances to protect foraging and loafing waterbirds from human disturbance in Florida. Wildl. Soc. Bull. 25(1):139-145.

Schenk, J. E., P. F. Atkins Jr., R. L. Weitzel, P. B. Simon, J. C. Posner, and W. J. Weber Jr. 1975. Effects of outboard marine engine exhaust on the aquatic environment. Prog. Wat. Tech. 7(3/4):733-741.

Stalmaster, M. V., and J. L. Kaiser. 1998. Effects of recreational activity on wintering bald eagles. Wildlife Monographs; No. 137, 46p.

U. S. Army Corps of Engineers (USACE). 1994. Cumulative impacts of recreational boating on the Fox River - Chain O' Lakes area in Lake and McHenry Counties, Illinois: Final Environmental Impact Statement. Environ. and Social Anal. Branch, U.S. Army Corps of Eng., Chicago, IL. 194 pp. + App.

Vermaat, J.E., and R. J. de Bruyne. 1993. Factors limiting the distribution of submerged waterplants in the lowland river Vecht (The Netherlands). Freshwater Biology 30:147-157.

Wagner, K. J. 1991. Assessing the impacts of motorized watercraft on lakes: Issues and perceptions. pp. 77-93. In: Proceedings of a National Conference on Enhancing States' Lake Management Programs, May 1990. Northeastern Illinois Planning Commission, Chicago IL.

Wagner, K. J. 1994. Of hammocks and horsepower: The noise issue at lakes. Lakeline, 14(2): 24-28.

Yousef, Y. A., W. M. McLellon, and H. H. Zebuth. 1980. Changes in phosphorus concentrations due to mixing by motor boats in shallow lakes. Water Research 14:841-852.

Zieman, J. C. 1976. The ecological effects of physical damage from motor boats on turtle grass beds in southern Florida. Aquatic Bot. 2:127-139.

10161

FINAL REPORT

SHIPSHEWANA LAKE RESTORATION FEASIBILITY STUDY

Submitted to:

SHIPSHEWANA COMMUNITY LAKE IMPROVEMENT ASSOCIATION

Submitted by:

INTERNATIONAL SCIENCE & TECHNOLOGY, INC.
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March 29, 1989

EXECUTIVE SUMMARY

Located in LaGrange County, Shipshewana Lake has a surface area of 202 acres, a mean depth of 6.7 ft., and a 4,675 acre watershed. Most of the drainage from the predominently agricultural watershed enters the lake through two ditches: Cotton Lake Ditch on the south shore, and the smaller Mud Lake Ditch to the west. Since the late 1970s, the water quality and general condition of Shipshewana Lake has declined dramatically. The Indiana Department of Environmental Management (IDEM) calculated an Indiana Lake Classification System (ILCS) Eutrophication Index (EI) value of 51 (on a scale of 0 to 75) for the lake (IDEM, 1986). Currently, Shipshewana Lake appears to be in the most extreme stages of eutrophication. All lake uses have been severely impaired; swimming has been banned, boating is hampered by algae scums and clumps of bottom material buoyed by gas bubbles, and fishing has declined precipitously.

The objectives of this feasibility study were to assess the current eutrophication-related problems of Shipshewana Lake, identify the sources of the problems and their relative contributions, and develop recommendations for a restoration program that will provide a reasonable probability of success in improving the quality of the lake.

The lake water quality data collected were characteristic of shallow productive lakes. An ILCS EI value of 53 was calculated from the data, indicating that there has been a deterioration in the lake's trophic status. Surficial sediments were observed to be primarily composed of silt/clay with a significant organic content. There was a distinctly flocculant layer at the surface of the lake bottom that appeared to consist largely of dead algae cells. This unconsolidated flocculant material represents a significant source of internal nutrient loading to the lake.

Cotton Lake and Mud Lake Ditches appear to be significant sources of sediment and nutrient loading to Shipshewana Lake. Agricultural and urban non-point source pollution is the principle source of nutrients entering the lake. A significant component of this pollution consists of animal wastes that are not being adequately controlled in the watershed. In addition, livestock access to streams is resulting in stream bank erosion and deposition and transport of wastes directly into the streams.

The urban area in and around the town of Shipshewana appears to be a significant source of nutrient contamination. Within this subbasin, a flea market, livestock auction house, and fertilizer plant are the most probable sources of significant discharges. There may be some leakage from the town sanitary sewer system or septic fields, as indicated by fecal coliform/fecal streptococcus ratios characteristic of human waste.

Restoration of Shipshewana Lake must include a significant reduction in external nutrient loading to the lake along with a reduction in available internal nutrients in the lake. Implementation of Best Management Practices (BMPs) throughout the watershed should begin with an extensive education program designed to make all farmers, businessmen, and residents aware of the issues surrounding the decline of Shipshewana Lake and the activities that result in nutrient contamination of runoff. The next step should be providing guidance in selection and implementation of appropriate practices, with emphasis on animal waste management, erosion control, and fertilizer contamination. Consideration should be given to construction of artificial wetlands, or similar treatment facilities at the mouths of the two ditches that discharge into the lake.

An over-winter lake drawdown is recommended to consolidate the loose organic sediments, reduce nutrient availability from these sediments, and provide some preventative action against the development of nuisance aquatic weeds as water quality improves in the future. Drawing the lake down by approximately 8 ft. will expose most of the organic sediments and allow a deep pool to remain for fish. Limited physical removal of sediment should be considered in certain areas where the organic material is especially thick. Installation of a water control structure at the outlet of the lake should also be considered.

It may be possible to take no in-lake actions and realize a significant long-term reduction in internal nutrients through washout. Under significantly reduced external loading, the lake will begin to export nutrients until a new equilibrium is reached between nutrient inputs and losses. The relatively short detention time of the lake (i.e., 2-3 months) should expedite this process. It is possible that this new trophic condition will be satisfactory for renewed use of the lake.

SECTION 1. INTRODUCTION AND BACKGROUND

1.1 SHIPSHEWANA LAKE AND WATERSHED

Located in LaGrange County, Shipshewana Lake has a surface area of 202 acres (Figure 1). The shores of the lake are heavily developed with single family residences and a youth camp. There is a state-owned public access site with a cement boat ramp on the south shore. The 4,675 acre watershed is predominantly agricultural, and most of the drainage enters the lake through two ditches: Cotton Lake Ditch on the south shore, and the smaller Mud Lake Ditch to the west. The lake discharges into Page Ditch through a sheet-pile dam on the eastern shore that was constructed in the early 1960s.

The lake has always been relatively shallow (i.e., mean depth of 6.7 ft. and maximum depth of 14 ft.), with a hard bottom of marl and sand. Aerial photographs taken in 1938 indicate that the lake was entirely surrounded by wetlands, except for the eastern shore which was forested (Grant 1988). At that time, Cotton Lake Ditch discharged into a large wetland area adjacent to the outlet of the lake. The watershed consisted of approximately 1,875 acres of cropland, with residential development limited to a few cottages on the southern-most shore of the lake.

By 1965 the eastern shore of Shipshewana Lake was fully developed with residential housing, and Cotton Lake Ditch had been diverted into the lake to provide lake access to land owners along the ditch. This diversion added 2,300 acres to the total watershed area, including 87 acres in the town of Shipshewana. Cotton Lake Ditch drains through two ponds (the Fish Ponds), the larger of which has a surface area of 5 acres, while the smaller pond is approximateky one acre.

The most recent available aerial photograph, taken in 1986, shows the development of the western shore of the lake where a marsh previously existed at the mouth of Mud Lake Ditch. The marsh was drained and filled with dredged material from adjacent marsh and the lake. In addition, several channels had been excavated around the lake to provide lake access to new development.

Significant features in the watershed include two facilities in the town of Shipshewana that draw upwards of 30,000 visitors and tourists to the area weekly between May and September: a cattle and horse auction facility, and a very large flea market. The town of Shipshewana has a sanitary sewer system. The sewage treatment plant discharges into Page Ditch, downstream of Shipshewana Lake. Stormwater runoff is by sheet flow, street gutters, and storm sewers that eventually discharge into Sarah Davis Ditch (Figure 1). The surrounding farming community is predominantly Amish, and farming techniques are typical of those commonly practiced in the late 1800s. Horse drawn vehicles are common and ubiquitous throughout the community.

1.2 NATURE OF THE PROBLEM

Since the late 1970s, the water quality and general condition of Shipshewana Lake has declined dramatically. The lake was classified as a Class Three lake (i.e., lowest quality, advanced eutrophic) by the Indiana Department of Environmental Management (IDEM) in the Indiana Lake Classification System and Management Plan (1986). This classification was based, in part, on a total phosphorus (TP) concentration of 0.045 mg/l and a Secchi disc transparency of 3.0 ft. A Eutrophication Index value of 51 was calculated for the lake, on a scale of 0 - 75. The lake was defined as being in the IV-A Lake Management Group: relatively shallow lakes that have the potential for being restored through dredging, bottom sealing, or sediment consolidation.

Shipshewana Lake appears to be in the most extreme stages of eutrophication. Massive blooms of blue-green algae have become common during the growing season. Secchi disc transparency has declined to approximately 0.5 ft. The hard lake bottom has been overlain by a thick and unconsolidated layer of organic sediment. Reducing (i.e., anoxic) conditions persist in this layer and in the water column above it during the peak growing season, as evidenced by total depletion of dissolved

SECTION 3. DISCUSSION

This section presents an evaluation and discussion of the data that were collected and reviewed for this study. The objective was to develop a clear definition of the current conditions in Shipshewana Lake and the mechanisms that have been responsible for these conditions.

3.1 TROPHIC INDEX ASSESSMENT

The Indiana Lake Classification System (ILCS) provides a mechanism for quantifying the current trophic status of Shipshewana Lake. An ILCS Eutrophication Index (EI) value of 53 was calculated for Shipshewana Lake using the water quality data collected during the field survey. Table 10 presents the details of the calculation.

There is one source of uncertainty in the EI calculation that should be noted. The samples for total organic nitrogen (TON) were destroyed in shipment. This is one of the parameters required for the index value calculation. Fortunately, the structure of the EI is such that a reasonably confident score may be estimated by inference from the other nitrogen constituents. Specifically, the mean NH_3 concentration was observed to be 0.85 mg/l. TON includes NH_3, plus other organic forms of nitrogen (e.g., particulate). Therefore, TON must be equal to or greater than 0.85 mg/l. Historical data (Table 1) indicate that total nitrogen, which is the sum of TON plus nitrates and nitrites, is typically 3.5 to 19 times greater than NH_3. Under these circumstances, the TON component of the EI has two possible scoring ranges: 0.9 - 1.9, and >1.9 mg/l. Conservatively, it may be confidently assumed that TON falls within the lower range, and this assumption was used in calculating the index value. The index value would have been one point higher, or 54, if TON were actually in the higher range.

An EI value of 51 was reported by IDEM (1986) for Shipshewana Lake. The current value of 53 indicates that there has been a deterioration in the lake's trophic status. This substantiates the qualitative

TABLE 10. ISBH LAKE EUTROPHICATION INDEX
Shipshewana Lake - 9/13/88

PARAMETER AND RANGE	RANGE VALUE	RANGE OBSERVED	POINTS
Total Phosphorus (mg/l)			
At least 0.03	1		0
0.04 - 0.05	2		0
0.06 - 0.19	3	x	3
0.2 - 0.99	4		0
Greater than 0.99	5		0
Soluble Phosphorus (mg/l)			
Less than 0.03	0	x	0
At least 0.03	1		0
0.04 - 0.05	2		0
0.06 - 0.19	3		0
0.2 - 0.99	4		0
Greater than 0.99	5		0
Organic Nitrogen (mg/l)			
At least 0.5	1		0
0.6 - 0.8	2		0
0.9 - 1.9	3	x	3
Greater than 1.9	4		
Nitrate (mg/l)			
At least 0.3	1		0
0.4 - 0.8	2		0
0.9 - 1.9	3	x	3
Greater than 1.9	4		0
Ammonia (mg/l)			
At least 0.3	1		0
0.4 - 0.5	2		0
0.6 - 0.9	3	x	3
Greater than 0.9	4		
Percent Saturation DO @ 5 ft.			
115	0		0
115 - 119	1	x	1
120 - 129	2		0
130 - 149	3		0
Greater than 149	4		0
Percent of water column with DO 0.1 ppm			
29	4		0
29 - 49	3		0
50 - 65	2		0
66 - 75	1		0
76 - 100	0	x	0

TABLE 10. ISBH LAKE EUTROPHICATION INDEX
Shipshewana Lake - 9/13/88 (concluded)

PARAMETER AND RANGE	RANGE VALUE	RANGE OBSERVED	POINTS
Secchi Transparency			
Five feet or less	6	x	6
Percent Light Transmission at 3 ft			
0 - 30	4	x	4
31 - 50	3		0
51 - 70	2		0
71 - 100	0		0
Total Plankton per ml			
Vertical tow from 5 ft.			
500	0		0
500 - 1,000	1		0
1,000 - 2,000	2		0
2,000 - 3,000	3		0
3,000 - 6,000	4		0
6,000 - 10,000	5		0
Greater than 10,000	10	x	10
Blue-Green Dominance?	5	x	5
Vertical tow from 5 ft. incl. thermocline			
1,000	0		0
1,000 - 2,000	1		0
2,000 - 5,000	2		0
5,000 - 10,000	3		0
10,000 - 20,000	4		0
20,000 - 30,000	5		0
Greater than 30,000	10	x	10
Blue-Green Dominance?	5	x	5
100,000	5		0
	Total Eutrophy Score		53

observations provided by residents of the surrounding community that the condition of the lake has declined.

3.2 HYDROLOGIC CONDITIONS

The principal hydrologic parameter of interest in evaluating the trophic status of Shipshewana Lake and developing a restoration strategy is hydraulic detention time. This is defined as the average time required for the volume of the lake to be completely replaced, and may be estimated from annual runoff, watershed area, and lake volume.

The U.S. Geological Survey has published average annual runoff data for the 48 coterminous states. The region containing LaGrange County has an average annual runoff of approximately 12.5 inches. Thus, the 4,625 acre Shipshewana Lake watershed may be expected to produce approximately 4,670 ac-ft. of runoff annually. In addition, the lake will receive approximately 38 in. of direct rainfall annually, and lose approximately 26.4 in. in evaporation. Based on a mean depth of 6 ft. and a surface area of 202 acres, Shipshewana Lake has a volume of approximately 1210 ac-ft. Thus, the lake's detention time was estimated to be approximately 0.24 yr., or about 88 days. This is a relatively short detention time.

Short detention times in lakes result in rapid flushing and retention of a relatively smaller fraction of total nutrient loading than would be retained in similar lakes with longer detention times (Reckhow, et al. 1979). From the perspective of lake restoration, this means that Shipshewana Lake is likely to have a relatively rapid response to a reduction in external nutrient loading. Moreover, accumulated sediment nutrients will be subject to reduction by flushing effects as released nutrients are washed out of the system and not replaced by external loads.

3.3 WATER QUALITY

The in-situ water quality data indicate that Shipshewana Lake was generally well mixed and not thermally stratified when sampled. The observed water column temperature and dissolved oxygen profiles are characteristic of relatively shallow productive lakes strongly influenced by wind mixing. Supersaturation of the water column reflects a highly productive algal population. The surficial organic sediments appear to be highly reduced and anoxic, as indicated by qualitative observations of the black color, sulfurous odor, and presence of gas bubbles in these sediments.

The nutrient concentrations observed in the water samples are characteristic of eutrophic systems. The mean TP concentration for in-lake stations (i.e., SL-1, SL-2, SL-3, and SL-4) was relatively high (0.090 mg/l) while the mean PO_4 concentration was near the detection limit (0.010 mg/l). These data indicate that most of the phosphorus in the water column was in particulate form, probably as algal biomass, and that phosphorus is the limiting nutrient. It is noteworthy that the recent historical data indicated TP concentrations significantly higher (0.38 mg/l in June 1988) than those observed during this investigation. Spatial and temporal variability in water quality parameters reflect the fundamental instability of eutrophic systems (Barica 1980). Seasonal and transient meteorological effects serve as driving influences on these lakes. Therefore, observed nutrient concentrations are expected to be inherently variable over relatively short periods of time.

The mean NH_3 and NO_3-NO_2 concentrations at the in-lake stations were 0.85 mg/l and 1.47 mg/l, respectively. Significant concentrations of these soluble forms of nitrogen indicate that nitrogen is not the limiting nutrient to algal growth in Shipshewana Lake. This substantiates the conclusion that phosphorus is the limiting nutrient in the lake.

The high nutrient concentrations observed at the stations along Cotton Lake Ditch indicate that this stream represents a significant source of phosphorus and nitrogen to Shipshewana Lake, even under low flow conditions. Several likely sources of nutrients were identified during the watershed reconnaissance, including livestock auction and flea market facilities in the town of Shipshewana, and a fertilizer plant adjacent to the Fish Ponds. However, runoff from these sources enters the inflow stream downstream of Cotton Lake. The very high levels of TP (0.2 mg/l) present in Cotton Lake and the emerald green color of the water observed during the site visit (indicating a concentrated bloom of bluegreen algae) imply a significant nutrient source upstream of Cotton Lake. Interestingly, the upstream station (CL-7) data showed relatively low TP and PO_4 concentrations, and relatively high NO_3-NO_2 and NH_3 concentrations. The sample appeared to be dominated by soluble forms of nitrogen; TSS and CHLA were very low. Much of the land upstream of Cotton Lake is agricultural, although there is also a mobile home camp within the subbasin. The data are insufficient to characterize the source of the contamination conclusively, but there appears to be a very significant source of nutrients in the upper reaches of the Cotton Lake subbasin.

Mud Lake Ditch is another significant source of nutrients to Shipshewana Lake, but primarily under storm flow conditions. At low flow conditions, the discharge from the ditch was negligible and Mud Lake (MD-4) had relatively low nutrient concentrations and suspended solids. In contrast, the storm samples collected in the lower reaches of the ditch on September 20 had very high nutrient and suspended solids concentrations. In particular, concentrations of the soluble nitrogen fractions (NH_3 and NO_3-NO_2) were the highest observed in any sample (4.80 mg/l and 14.0 mg/l, respectively).

Although the base flow fecal coliform and fecal streptococcus cell counts were relatively low, the ratios of fecal coliform to fecal streptococcus (FC/FS) may provide information on possible sources of pollution (Table 6). A ratio greater than 4.1 is considered indicative of pollution derived from domestic wastes (human) whereas ratios less than 0.7 suggest pollution due to nonhuman sources (Metcalf & Eddy 1979). Ratios between 0.7 and 4.4 usually indicate wastes of mixed human and animal sources. Using these values as a guide, the Cotton Lake inflow would appear to be affected primarily by human wastes in and immediately downstream of the Fish Ponds and in the upper part of the watershed. Those portions of the stream between Cotton Lake and the Fish Ponds (including Sarah Davis Ditch) appear to be contaminated by a mixture of human and animal wastes. Because of the low cell counts involved in calculating the ratios, any inferences must be interpreted as being suggestive, but not conclusive.

The Mud Lake Ditch discharge FC/FS ratios indicate that significant contamination is occurring under storm flows, probably from poultry wastes. It is significant that the total numbers of fecal coliform and fecal streptococcus were very high at the MD-1 station on October 18.

3.4 ALGAE POPULATIONS

The phytoplankton data indicate that Shipshewana Lake has a highly productive algae population, dominated by blue-green species. This is typical of highly eutrophic systems. Although the observed water column data indicate nearly complete oxygenation throughout the water column, the presence of such a productive and diverse algal community presents a serious threat of a population bloom-collapse cycle (Mericas, et al 1985). Under optimum environmental and meteorological conditions, such as a prolonged period of clear skies, warm temperatures, and calm winds, algae populations reach very high concentrations which are not sustainable under normal conditions. When conditions change, with the passage of a weather system for example, the unstable population collapses, resulting in a near-complete deoxygenation of the water column as the algal biomass decomposes. In addition to the impacts on aquatic organisms, these bloom-collapse events present aesthetic problems such as odors and unsightly scums. Although there is no evidence of toxic effects in Shipshewana Lake, certain blue-green species, including Anabaena sp. and Aphanizomenon flos-aquae present the potential for toxic effects.

3.5 LAKE SEDIMENTS

The lake sediment core visual and analytical data indicate that the surficial sediments in Shipshewana Lake are primarily composed of silt/clay with a significant organic content. The cores taken at SL-1, SL-2, and SL-4, had a mean organic content of 34.8 percent. This is on the lower end of the range of organic content reported by Wedepohl et. al. (1983) for seven glacial lakes in Wisconsin. There was a distinctly flocculant layer at the surface of each of these cores that appeared to consist largely of dead algae cells. This type of unconsolidated flocculant material typically contains large concentrations of nitrogen and phosphorus.

The very low organic content observed at the SL-3 station is probably indicative of a spring in that area of the lake. The heterogeneity of organic sediment in the lake is evident from the variability among stations. In addition, a comparison of the location of SL-3 and the organic sediment depth map (Figure 8) indicates that substantial differences may occur within close proximity. Again, springs in the bottom of the lake would account for this variability. Unfortunately, precise definition of the locations and flow of springs in the lake was beyond the scope of this investigation.

The observed characteristics of the surficial sediments in Shipshewana Lake indicate that they are probably a significant source of internal nutrient loading, especially under the anoxic conditions that appear to predominate at the sediment-water interface. With a significant reduction in external nutrient loading, this internal nutrient source may be sufficient to sustain eutrophic conditions, albeit less severe than currently exist in the lake. Any restoration effort must address the problem of significant sediment nutrient release.

3.6 LAND USE

The review of land use in the watershed revealed several specific activities and general practices that represent potential sources of nutrient loading to Shipshewana Lake. The identification of these specific nutrient sources is based on the nature of the activities at each location and literature data on nutrient loading associated with those types of land use. Sampling and quantification of specific nutrient sources was beyond the scope of this investigation, and is not anticipated as being required to define a restoration strategy. However,

SECTION 5. CONCLUSIONS AND RECOMMENDATIONS

5.1 CONCLUSIONS

Shipshewana Lake is suffering from excessive nutrient enrichment and is rapidly moving from eutrophic to hypereutrophic conditions. The evidence for this lies in elevated nutrient concentrations found in the lake, the increasing frequency and severity of algal blooms, the dominance of blue-green algae in these blooms, and the observed increase in the lake's Eutrophication Index score. A significant layer of flocculant organic sediment is resulting in accelerated nutrient cycling between the sediments and water column, and generally anaerobic conditions on the lake bottom.

Agricultural and urban non-point source pollution is the principle source of nutrients entering the lake. A significant component of this pollution consists of animal wastes that are not being adequately controlled in the watershed. The most severe individual animal waste problem observed was at the duck farm on Mud Lake Ditch where the existing waste settling tank was observed to be full to capacity. In addition, livestock access to streams is resulting in stream bank erosion and deposition and transport of wastes directly into the streams.

The urban area in and around the town of Shipshewana appears to be a significant source of nutrient contamination. Within this subbasin, the flea market, livestock auction house, and fertilizer plant appear to be the most probable sources of significant discharges. In addition, there may be some leakage from the town sanitary sewer system or septic fields, as indicated by fecal coliform/fecal streptococcus ratios.

Restoration of Shipshewana Lake must include a significant reduction in external nutrient loading to the lake along with a reduction in available internal nutrients in the lake. This may be accomplished through an integrated program of implementation of agricultural and urban best management practices throughout the watershed and mitigation measures in the lake. Of the in-lake alternatives, the most economical and likely to succeed will be a drawdown to promote sediment consolidation and inhibit nuisance macrophyte development. Limited mechanical dredging may be possible while the lake level is down, and will be beneficial to the lake.

It may be possible to take no in-lake actions and realize a significant long-term reduction in internal nutrients through washout. Under significantly reduced external loading, the lake will begin to export nutrients until a new equilibrium is reached between nutrient inputs and losses. It is possible that this new trophic condition will be satisfactory for renewed use of the lake.

5.2 RECOMMENDATIONS

A restoration of Shipshewana Lake is warranted by the current eutrophic conditions and the impairment to recreational uses that these conditions have caused. The restoration should include a combination of reduction in external nutrient load and internal nutrient cycling.

Implementation of BMPs throughout the watershed should begin with an extensive education program designed to make all farmers, businessmen, and residents aware of the issues surrounding the decline of Shipshewana Lake and the activities that result in nutrient contamination of runoff. Once the population has been made aware of the nature of the problem, the next step should be providing guidance in selection and implementation of appropriate management practices. Emphasis should be placed on animal waste management, erosion control, and fertilizer contamination. This program should be conducted in conjunction with the County Conservationist's office and the SCS.

Wetland areas historically existed at the mouths of Cotton Lake Ditch and Mud Lake Ditch. Serious consideration should be given to construction of artificial wetlands at the mouths of these two discharges. Such wetlands will serve to further reduce sediment and nutrient loading to the lake through treatment of the ditch discharges.

In-lake restoration efforts should only begin after the implementation of controls in the watershed. Premature in-lake mitigation efforts are likely to be undone by continued loading from the watershed. An over-winter lake drawdown is recommended to consolidate the loose organic sediments, reduce nutrient availability from these sediments, and provide some preventative action against the development of nuisance aquatic weeds as water quality improves in the future. Drawing the lake down by approximately 8 ft. will expose most of the organic sediments and allow a deep pool to remain for fish.

A certain amount of dredging will be required to provide a drainage channel between the deepest part of the lake and the lake outlet. While the excavation machinery is on-site for this dredging, additional limited physical removal of sediment should be considered in accessible areas where the organic material is especially thick. An additional construction activity that should be considered is the installation of a water control structure at the outlet of the lake that would allow periodic drawdowns in the future for lake maintenance (e.g., weed control, fisheries renovations, and dredging).

5.3 FUTURE ACTIONS

Consistent with the structure of the Indiana Lake Enhancement Program, the next step in the restoration process should be a Design Study, followed by implementation of the restoration activities.

5.3.1 Design Study

A Design Study should be conducted with the following objectives:

- Identify and design site specific BMPs for the major sources of nutrient contamination in the watershed (i.e., livestock auction, flea market, fertilizer plant, livestock holding areas). This should be done in close cooperation with the District Conservationist and the SCS.

- Design artificial wetlands, sediment traps, or similar treatment systems for the discharges of Cotton and Mud Lake Ditches.

- Design and plan a lake drawdown to provide a reduction in lake level of approximately 8 ft. This plan should include specifications for the excavation required for adequate drainage, opportunistic dredging that might be conducted in association with the drawdown, disposal of the excavated material, construction of an outlet structure to facilitate future drawdowns, and permits that may be required.

- Conduct an environmental impact assessment of downstream effects of an extended drawdown of Shipshewana Lake.

- Investigate funding options available to support the restoration implementation.

- Prepare and submit application packages for permits and funding.

5.3.2 Restoration Implementation

The restoration effort should begin with implementation of BMPs for controlling contaminated runoff throughout the watershed. It will probably take several years for most of the individual landowners, businessmen, and farmers to construct on-site controls. During this period all regional facilities, such as detention ponds, wetlands, and sediment traps should be constructed.

A drawdown of Shipshewana Lake should be conducted after most of the watershed controls have been put into operation. The drawdown should be scheduled to begin in September or October to allow the lake to fully drain and the exposed sediments to dry prior to the onset of Winter snow and ice cover. All excavation, dredging, and construction should be scheduled for this period as well. The lake should be allowed to refill in the following Spring. The available hydrologic information indicates that the lake will refill within 2 to 3 months.

THE OHIO STATE UNIVERSITY EXTENSION

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Planktonic Algae in Ponds

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Many pond owners are familiar with the problems caused by filamentous algae. The stringy, fast-growing algae can cover a pond with slimy, lime-green clumps or mats in a short period of time. Fortunately, products are available to assist the owner in controlling filamentous algae (Ohio State University Extension Fact Sheet A-3-98 *Controlling Filamentous Algae in Ponds*). Planktonic algae are another group of algae common to ponds. These algae are critical to a pond's food chain as they provide food for the microscopic animals that in turn are eaten by freshly-hatched fish fry. Occasionally, planktonic algae can "bloom" to nuisance levels which may necessitate using control methods.

What are Planktonic Algae?

These algae are microscopic free-floating plants. They are suspended in the top few feet of water of a pond or lake where light is bright enough for them to produce food by photosynthesis. The planktonic algae community in ponds is typically composed of green algae, blue-green algae, diatoms, and euglenas. Some species of planktonic algae, primarily blue-greens, can be toxic to animals and impart an odor or taste to the water.

Seasonal Abundance

Planktonic algae are least abundant in winter when cold water temperatures inhibit their reproduction and growth. This is why most ponds are their clearest in winter. As ponds warm in April, reproduction by algae increases greatly and the spring algae "bloom" occurs. The pond's water becomes distinctly less clear, with water color becoming greenish or brownish depending on the algae species present. In late April and May, reproduction by various microscopic animals (rotifers and daphnia, for example) creates large populations of these animals which begin to "crop" the algae bloom. The water becomes clearer. Once water temperatures reach about 72 degrees F, the microscopic animal population declines rapidly through decreased reproduction and predation by small fish fry. This allows the planktonic algae to rebound, but usually not to levels of April. In most ponds, planktonic algae levels remain stable throughout the summer unless there is a sudden, unexpected source of new nutrients to cause a "summer bloom." As ponds cool in fall, abundance slowly declines to winter levels. Many ponds become noticeably clearer during fall as algae abundance declines.

When are Planktonic Algae a Problem?

Planktonic algae blooms are rarely a problem for Ohio pond owners, but when they do occur the blooms cause considerable concern. Algae blooms cause the water to appear pea soup green or brown in color. Severe blooms often create the appearance of green paint being spilled on the pond's surface. For the pond owner who desires a clear pond for swimming or for aesthetic reasons, this situation is unacceptable.

Severe algae blooms can cause problems for fish. If the bloom dies-off suddenly, a fish summer kill can result due to oxygen depletion. Bloom die-off can be caused by weather changes, a sudden decrease in nutrient levels, or treatment of the pond with an herbicide. Additional information on fish summer kill can be found in Ohio State University Extension Fact Sheet A-8-01 *Winter and Summer Fish Kills in Ponds*.

What Causes a Nuisance Bloom?

Like all plants, algae require nutrients to grow and reproduce. Because algae are free-floating, they must get those nutrients from the water. They have no ability to obtain them from the pond bottom. Thus, the higher the nutrient level in the pond, the more algae you will have. At slightly higher nutrient levels, the algae community is often dominated by filamentous algae. This is particularly true during summer. At very high nutrient levels, the algae blooms are typically composed of planktonic algae rather than filamentous algae. Fortunately, few Ohio ponds have very high nutrient levels unless they are receiving unwanted nutrients from the pond's watershed.

There are many possible sources of excess nutrients that might enter a pond and cause a nuisance bloom. Some of the more common sources are fertilizing grass around ponds, too many geese, run-off from livestock operations, and leaking septic systems. Another common cause of summer planktonic algae blooms is the sudden release of nutrients from dying aquatic vegetation or filamentous algae that has been treated with an herbicide. Again, the more excess nutrients added, the more likely the resulting bloom will reach nuisance levels.

Prevention and Treatment

Prevention—Preventing an algae bloom is always preferable to the expense of treating with an algaecide to eliminate it. If a pond owner wishes to avoid a nuisance bloom, he or she should investigate the pond's watershed and determine potential sources of excess nutrients. If sources are found, then a plan to eliminate the source should be developed and implemented. This may require working closely with neighbors.

If a pond is experiencing a severe nuisance algae bloom, the owner should perform the above investigation but may want to postpone implementation until winter if fish are important. As mentioned previously, the sudden depletion of nutrients to a pond with a severe ongoing algae bloom could cause a summer fish kill. In this case, it makes sense to allow cooler fall weather to decrease the algae bloom and then implement a plan to prevent new blooms the next year.

In some situations, it may be impossible to eliminate or substantially reduce unwanted nutrients from entering a pond. Uncooperative neighbors is a common reason for this. A preventative control method is the use of an aquatic dye, such as Aquashade, to prevent initial growth early in spring. The dye must be added by April 1 to insure prevention of that year's algae blooms. A drawback to the use of dyes is that reducing the production of planktonic algae impacts the pond's food chain. Less production at the bottom of the food chain can mean fewer pounds of fish in the pond. Many owners of very nutrient-rich ponds consider this an acceptable outcome when using the dyes.

Algaecide—A nuisance planktonic algae bloom can be quickly reduced with the use of liquid chelated copper compounds, such as Cutrine-Plus or Algae-Pro. Again, it is important to note that a sudden summer die-off of algae can cause a fish kill. Planktonic algae have a very high reproductive rate, so re-bloom may occur in just a few weeks following treatment. Several treatments may be necessary for seasonal control.

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CHANGES IN PHOSPHORUS CONCENTRATIONS DUE TO MIXING BY MOTORBOATS IN SHALLOW LAKES

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Abstract—Recreational motorboats equipped with engines varying from 28 to 165 horsepower were operated at three selected Central Florida lakes. Also, a pair of isolation chambers representing aquatic habitats were placed in each lake for control and mixing studies. Mixing in isolation chambers was performed by small electrical motors connected to two blade propellers.

Agitation of the water column in the lake mixing stations and inside the isolation chambers increased water turbidities and phosphorus concentrations. The increase in turbidity and phosphorus content occurred at a much higher rate than the rate of decline after cessation of mixing. The increase in phosphorus content could result in an increase in lake productivity as noticed from the increase of chlorophyll *a* concentrations in lake mixing stations.

INTRODUCTION

Phosphorus has usually been considered the limiting nutrient in most aquatic systems (Fitzgerald, 1970; Syers, 1973, and USEPA, 1976). The amount of phosphorus present in most aquatic organisms has been found to be only a small percentage of their total weight. However, phytoplankton can absorb more than ten times their normal content of phosphorus and store it for future use. Zooplankton and littoral macrophytes also absorb and store orthophosphate directly from the water column. In environments rich in phosphorus, the luxury phosphorus is usually stored in the form of polyphosphates which have been found to comprise up to 20% of the cell dry weight (Hooper, 1973).

The suspended plankton in a water column settle and become part of the sediment. Sediment phosphorus content has been observed to be generally highest near the sediment–water interface and decreasing with depth as illustrated by sediment phosphorus profiles in several Florida lakes (Stewart, 1976). Phosphorus in the sediment is present in both the soluble and insoluble states (Syers *et al.*, 1973; Williams & Mayer, 1972; Williams *et al.*, 1971).

Mixing of the sediments and the overlying water occurs due to natural and man-made processes. Principal processes of natural mixing have been found to result from thermal gradients, wind and pressure waves, mobility of organisms, such as insect larvae and worms, and the formation of gas bubbles and pockets, either methane, carbon dioxide, or oxygen. Man-made mixing results from recreational activities such as boating and artificial mixing of the lake. Mixing of the sediments and the overlying water would have a number of effects on the phosphorus concentrations and forms present in the sediment. Interstitial water, containing up to fifty times the concentration of soluble ortho-PO_4, would be mixed with the overlying water. After resettling, phosphorus contained in lower sediments and interstitial waters could diffuse upward, replacing the supply of orthophosphate lost to the water column. It has been reported that 19–65% of the inorganic phosphorus absorbed by bottom sediments is exchangeable (Li *et al.*, 1973). Continued mixing would increase the rate of phosphate exchanged (Kuo & Lotse, 1974). Also, contact with aerobic lake water would cause some ortho-PO_4 to be lost to the water column with the conversion of ferrous hydroxide associated with sediments to ferric oxyhydroxide and the resultant decrease in its adsorptive capacity. In addition, bacteria on the now aerobic particles would exhibit a rapid uptake of inorganic phosphorus. Soluble organic phosphorus is released from both phytoplankton and bacteria (Lean, 1973; Hays, 1958). The rapid conversion of sediment orthophosphate to organic phosphorus in a water column is supported by Carter *et al.* (1974) studies.

Little work has been done to determine what role, if any, the increased use of outboard motors has played in the increased rate of eutrophication of many lake systems. Casey *et al.* (1974) and Neilson (1974) were interested in the rate of transfer of oxygen from

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Table 4. Average concentrations for selected water quality parameters collected from ICAH stations on study lakes

Parameter	Statistical variable	Lake Claire Control chamber Before	Lake Claire Control chamber After	Lake Claire Mixing chamber Before	Lake Claire Mixing chamber After	Lake Mizell Control chamber Before	Lake Mizell Control chamber After	Lake Mizell Mixing chamber Before	Lake Mizell Mixing chamber After	Lake Jessup Control chamber Before	Lake Jessup Control chamber After	Lake Jessup Mixing chamber Before	Lake Jessup Mixing chamber After
Turbidity	# of obs.	10	8	10	10	8	8	8	8	14	14	17	16
	Mean (JTU)	2.08	2.02	2.42	5.11	3.51	3.69	2.73	3.34	10.7	11	13.0	62.6
	Std. Dev.	0.66	0.40	0.55	3.54	0.97	1.17	0.92	1.09	7.91	4.6	6.12	45.3
Orthophosphorus	# of obs.	11	9	9	9	14	16	13	15	12	12	13	13
	Mean (mg l^{-1}-P)	0.033	0.030	0.031	0.044	0.027	0.031	0.029	0.0237	0.12	0.13	0.12	0.34
	Std. Dev.	0.009	0.011	0.012	0.14	0.010	0.010	0.006	0.014	0.04	0.05	0.03	0.24
Total phosphorus	# of obs.	11	12	11	11	16	16	16	16	12	12	13	13
	Mean (mg l^{-1}-P)	0.045	0.045	0.048	0.092	0.069	0.062	0.058	0.069	0.325	0.340	0.350	0.905
Chlorophyll *a*	# of obs.	14	13	14	13	18	17	18	17	15	15	17	9
	Mean (mg m^{-3})	5.6	5.3	5.2	8.8	14.4	18.1	5.1	7.4	65.9	77.2	68.5	99
	Std. Dev.	3.0	2.1	1.5	3.5	9.1	10.5	3.2	4.2	25.6	30.9	23.5	22.1

Table 5. Analysis of variance for selected water quality parameters collected before and after mixing periods

Lake	Station	Chance probability for Turbidity	Orthophosphorus	Total phosphorus	Chlorophyll *a*
Claire	Control	0.71	0.34	0.07	0.34
	Mixing	0.02	0.0001	0.003	0.005
	ICAH Control	0.82	0.93	0.62	0.79
	ICAH Mixing	0.07	0.041	0.08	0.002
Mizell	Control	0.15	0.60	0.70	0.29
	Mixing	0.21	0.13	0.002	0.26
	ICAH Control	0.74	0.27	0.46	0.28
	ICAH Mixing	0.24	0.10	0.45	0.09
Jessup	Control	0.64	0.83	0.77	0.70
	Mixing	0.008	0.001	0.008	0.12
	ICAH Control	0.85	0.58	0.77	0.26
	ICAH Mixing	0.001	0.002	0.005	0.002

Table 6. Phosphorus change due to mixing by motorboats in study lakes

Lake	Phosphorus form	Control stations *No. of Samples	Av. Conc ($mg\ l^{-1}$-P) Initial	Final	% change	Mixing stations *No. of Samples	Av. conc ($mg\ l^{-1}$-P) Initial	Final	% change
Claire	OP	42/40	0.039	0.041	5	72/69	0.042	0.060	43
(76–77)	TP	42/40	0.059	0.076	29	71/68	0.085	0.118	39
Mizell	OP	20/20	0.036	0.038	6	60/60	0.044	0.051	16
(1977)	TP	20/20	0.053	0.055	4	60/60	0.061	0.078	28
Jessup	OP	12/12	0.110	0.112	2	12/12	0.091	0.161	73
(1976)	TP	12/12	0.327	0.312	−5	12/11	0.304	0.470	55

* The top number represents samples collected before mixing and the lower number samples after mixing.

est for better understanding the mechanism and consequences involved. A rapid increase in OP–P to maximum value was reached after a few minutes of mixing in the isolation chamber (IC2) in Lake Jessup as shown from Fig. 6. The maximum turbidity and OP–P measurements occurred during the first 10 min of mixing time. A slight decline from the maximum value in turbidity measurements was noticed due to decline in the battery strength powering the electric motors used for mixing in the isolation chambers.

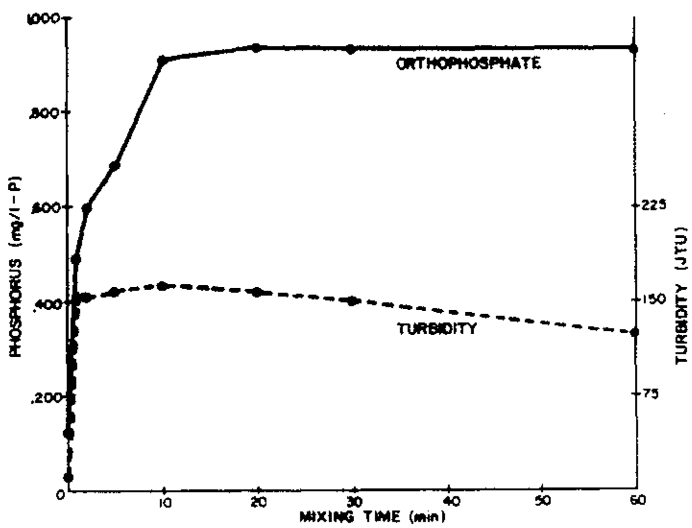

Fig. 6. Change of turbidity and phosphorus concentration with mixing time in chamber at Lake Jessup.

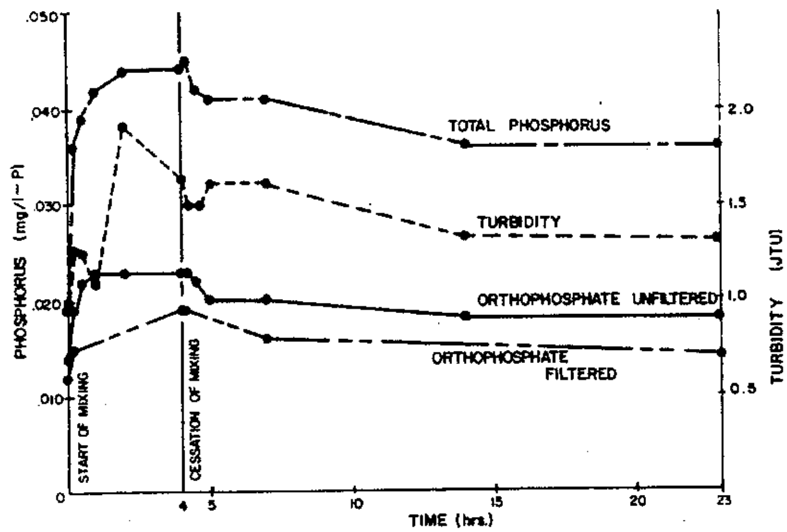

Fig. 7. Change of phosphorus concentration in mixing chamber at Lake Claire.

Phosphorus seems to increase at a much faster rate than the rate of decline after cessation of mixing. This was clearly demonstrated in Figs 7, 8 and 9. The maximum concentrations of phosphorus were reached in less than two hours of mixing in isolation chambers. Twenty hours after cessation of mixing the phosphorus concentrations, including OP, TP, filtered and unfiltered phosphorus remained higher than the initial phosphorus concentrations before mixing. The orthophosphorus rate of increase due to mixing averaged 0.041 $mg\,l^{-1}$-min, 0.009 $mg\,l^{-1}$-h, and 0.005 $mg\,l^{-1}$-h for Lakes Jessup, Claire and Mizell, respectively. The rates of decline in the orthophosphorus concentration over the first day after cessation of mixing averaged 0.011 $mg\,l^{-1}$-h, 0.0006 $mg\,l^{-1}$-h and 0.0003 $mg\,l^{-1}$-h for Lakes Jessup, Claire and Mizell. The dissolved orthophosphates averaged 81, 89 and 73% of the unfiltered OP–P for Lakes Claire, Mizell and Jessup, respectively.

PHOSPHORUS RELEASE STUDIES

Core samples of bottom sediments were collected and the upper 2 cm of sediment core were placed in an acid washed 4 l glass bottle, transported to the laboratory, stored at 4°C, and processed within 4 h. Each lake was sampled in triplicate from randomly selected sites in the mixing areas to investigate the leachable orthophosphorus from the sediments. A fourth sediment core was taken from the same mixing area in each lake to determine moisture, volatile solids and total phosphorus content of the sediment.

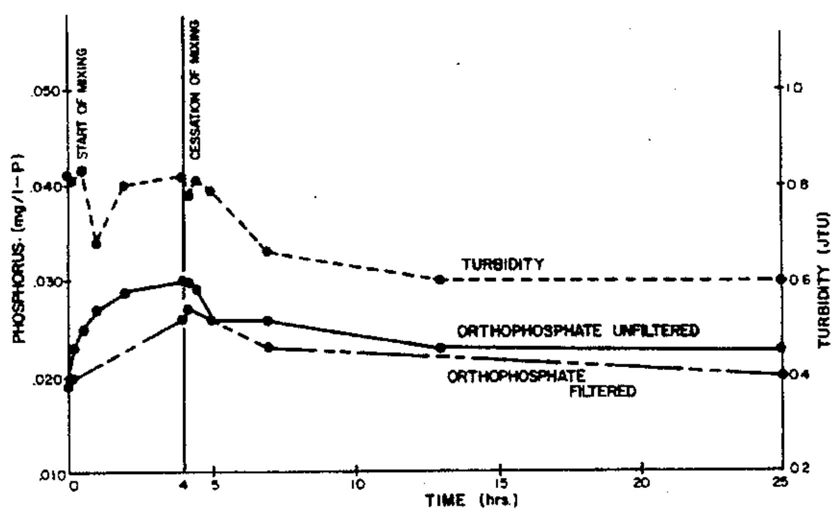

Fig. 8. Change of turbidity and phosphorus concentration in mixing chamber at Lake Mizell.

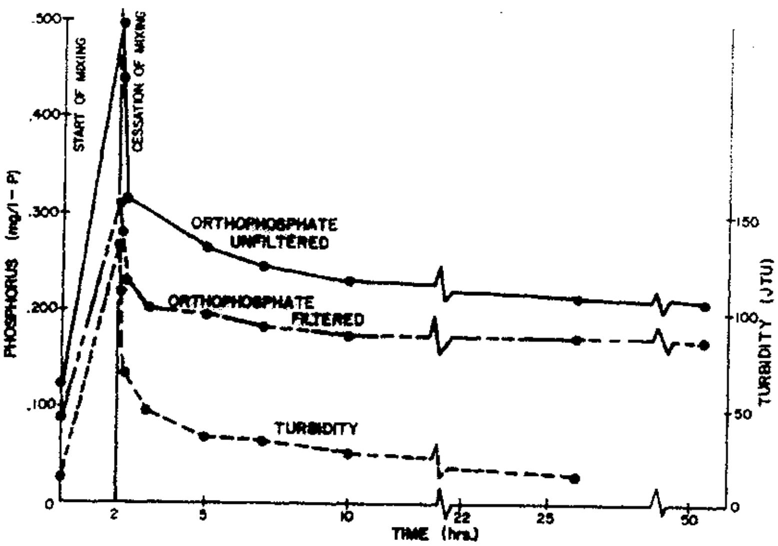

Fig. 9. Change of turbidity and phosphorus concentration in mixing chamber at Lake Jessup.

All tests were run in triplicates. A summary of the results is presented in Table 7. There was no difference between OP released after 5 min and 3 h contact time. Also, the OP released was highest from Lake Jessup sediments followed by Lake Claire and lowest from Lake Mizell. Filtered OP released was estimated to be 163, 74, and 41 mg m^{-2} from Jessup, Claire and Mizell bottom sediments, respectively. The phosphorus released from batch experiments followed the same general trend of phosphorus changes due to boating activities in Lakes Jessup, Claire and Mizell.

PHOSPHORUS–TURBIDITY RELATIONSHIPS

Regression analysis was performed to detect if correlations exist between turbidities and the phosphorus

Table 7. Summary of bottom sediment orthophosphorus release in water environment

Parameters	Lake Claire Mean	Lake Claire Std. Dev.	Lake Mizell Mean	Lake Mizell Std. Dev.	Lake Jessup Mean	Lake Jessup Std. Dev.
% Water content in bottom sediments core	30.9	4.3	24.7	0.20	90.7	0.5
% Volatile solids in bottom sediments core	2.5	1.8	0.76	0.16	33.7	1.6
Total phosphorus content (mg m^{-2})	5227	—	9456	—	7991	—
Leachable orthophosphorus						
A. Filtered (mg m^{-2})	74	31	41	16	163	75
B. Unfiltered (mg m^{-2})	123	20	53	19	194	68

Table 8. Correlations between water quality parameters for water samples collected from study lakes

Lake	Variable Independ. Var	Variable Depend. Var.	No. of obs n	Correl. coef. r	r values at 1% level of significance
Claire	Turb. (JTU)	OP (mg l^{-1}-P)	66	0.45	0.31
(1976)	Turb. (JTU)	TP (mg l^{-1}-P)	62	0.45	0.32
Claire	Turb. (JTU)	OP (mg l^{-1}-P)	174	0.42	0.19
(1977)	Turb. (JTU)	TP (mg l^{-1}-P)	174	0.38	0.19
Claire	Turb. (JTU)	OP (mg l^{-1}-P)	240	0.46	0.17
(1977 & 1976)	Turb. (JTU)	TP (mg l^{-1}-P)	236	0.52	0.17
Jessup	Turb. (JTU)	OP (mg l^{-1}-P)	96	0.78	0.26
(1976)	Turb. (JTU)	TP (mg l^{-1}-P)	96	0.82	0.26

content in samples collected from control and mixing stations of the study lakes, considering 1976 and 1977 data separately and combined. Positive correlations existed as shown in Table 8. All the calculated correlation coefficients r listed in this table are greater than the critical r values at a 1% level of significance. It appears that changes in turbidity measurements are accompanied by changes in the phosphorus content. In other words, if turbidity changes by induced waves from motorboats, it will produce changes in OP and TP. Also, if changes in turbidities could be minimized, changes in the phosphorus content could also be kept at a minimum. The engine/boat combination that limits the frequent disturbance of the bottom sediments in shallow lakes can be determined as suggested by Yousef *et al.* (1978).

SUMMARY AND CONCLUSIONS

Changes in water quality due to mixing by motorboats were studied in shallow lakes of Central Florida, namely Lakes Claire, Mizell and Jessup. The lakes differ in average water depth, sediment characteristics and trophic state.

The results showed significant increase in turbidity and ortho- and total phosphorus concentrations in water samples collected after mixing by motorboats in Lakes Claire and Jessup. The results obtained from Lake Mizell were not as conclusive. Lake Mizell was substantially deeper, had a sandy bottom sediment in the shallower depths and it was frequently used for recreation activities which made it difficult to maintain a valid control. The increase in the orthophosphorus content for Lakes Claire, Mizell and Jessup averaged 43, 16 and 73% which corresponded to an average of 46, 24 and 105 mg–P of orthophosphorus resuspended per square meter of the bottom sediments, respectively. Similarly, the increase in the total phosphorus content for Lakes Claire, Mizell and Jessup averaged 39, 28 and 55%, which corresponded to an average of 84, 58 and 249 mg–P m^{-2}, respectively.

Positive correlations existed between turbidities and the phosphorus content in the water column and the calculated correlation coefficients were greater than those required for a 1% level of significance. The rate of increase in the phosphorus content with mixing time is much higher than the rate of decline after cessation of mixing.

Data indicate substantial water quality effects are possible due to recreational boating on shallow lakes. These effects are significant and should be considered in water resource studies. Whether physical control of boating sizes and/or horsepower is required for these type lakes is not determined. However, the study provides a base for regulatory control to prevent degradation of water quality due to mixing and for consideration in waste load allocation modeling. Widespread interest in the results has developed both from citizens groups and governmental agencies. These interests are applicable to estuarine as well as fresh waters.

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REFERENCES

American Public Health Association (1975) American Water Works Association and Water Pollution Control Federation. *Standard Methods for the Examination of Water and Wastewater*, (14th edition).

Brezonik, P. L. (1972) Nitrogen: sources and transformations in natural waters. *Nutrients in Natural Waters*, (Edited by Allen H. E. & Krames J. R.) pp. 1–50. John Wiley & Sons, New York.

Brezonik P. *et al.* (1976) Analysis of eutrophication and water quality factors in the middle St. Johns river basin. Report No. ENV-07-76-02, Florida Department of Environmental Regulation, Tallahassee.

Carter D. L., Brown M. J., Robbins C. W. & Bondurant J. A. (1974) Phosphorus associated with sediments in irrigation and drainage waters for two large tracts in Southern Idaho. *J. Environ. Qual.* **3**, 287–291.

Casey T. J. & Karmo O. T. (1974) The influence of suspended solids on oxygen transfer in aeration systems. *Water Res.* **8**, 805–811.

Fitzgerald G. P. (1970) Aerobic lake muds for the removal of phosphorus from lake waters," *Limnology and Oceanography*, 15, pp. 550–555.

Hays F. R. & Phillips J. E. (1958) Lake water and sediment, radiophosphorus equilibrium with mud, plants and bacteria under oxidized and reduced conditions. *Limnol. Oceanogr.* **3**, 449–457.

Hooper F. F. (1973) Origin and fate of organic phosphorus compounds in aquatic systems. *Environmental Phosphorus Handbook*, (Edited by Griffith E. J., Beeton A., Spencer J. M. & Mitchell D. T.). John Wiley & Sons, New York.

Kuo S. & Lotse D. G. (1974) Kinetics of phosphate adsorption and desorption by lake sediments. *Soil Sci. Soc. Am. Proc.* **38**, 50–54.

Lean D. R. S. (1973) Movement of phosphorus between its biologically important forms in lake water. *J. Fish. Res. Bd Can.* **30**, 1525–1536.

Lee G. F. (1970) Factors affecting the transfer of materials between water and sediments. Office of Water Resources Research Project W-117 (#1614) (1), Water Chemistry Program, University of Wisconsin, Madison, Wisconsin, 1970.

Li W. C., Armstrong D. E. & Harris R. F. (1973) Measurements of exchangeable inorganic phosphate in lake sediments. *Environ. Sci. Technol.* **7**, 454–456.

Neilson B. J. (1974) Reaeration dynamics of reservoir destratification. *Am. Wat. Wks Ass.* **66**, 617.

Stewart E. A. (1976) A study of differences in vertical phosphorus profiles within the sediments of selected Florida lakes as related to trophic dynamics. M.S. Thesis, Civil Engineering and Environmental Sciences Department, Florida Technological University (now the University of Central Florida), Orlando.

Syers J. K., Harris R. F. & Armstrong D. E. (1973) Phosphate chemistry in lake sediments. *J. Environ. Qual.* **2**, 1–14.

U.S. Environmental Protection Agency (1974) *Methods for Chemical Analysis of Water and Wastes*, EPA-625/6-74-003, Office of Technology Transfer, Washington, D.C.

U.S. Environmental Protection Agency (1976) *National*

Set-Back Distances to Protect Nesting Bird Colonies from Human Disturbance in Florida

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Abstract: *Breeding colonial waterbirds are particularly susceptible to human disturbance because of their high-density nesting habits. Identified detriments to reproductive success include egg and nestling mortality, nest evacuation, reduced nestling body mass and slower growth, premature fledging, and modified adult behaviors. Fifteen species of colonial waterbirds nesting at 17 colonies in north and central Florida were exposed to three different human disturbance mechanisms (HDMs) in order to determine recommended set-back (RS) distances for protecting these mixed-species nesting assemblages. Both intraspecific and interspecific variation were observed in flushing response distances to the same human disturbance mechanisms. In general, colonial waterbirds exhibited greater average flush distances in reaction to a walking approach than to approaching motor boats. Recommended set-back distances were estimated using a formula based on the mean plus 1.6495 standard deviations of the observed flushing distances plus 40 meters* [$\hat{RS} = \exp(\hat{\mu} + 1.6495\hat{\sigma} + 40)$]. *In general, a recommended set-back distance of about 100 meters for wading bird colonies and 180 meters for mixed tern/skimmer colonies should be adequate to effectively buffer the sites we studied from human disturbance caused by approach of pedestrians and motor boats. We recommend follow-up studies to test our model at other breeding colonies.*

Distancia de alejamiento para proteger de las perturbaciones humanas a las colonias de aves nidificadoras en Florida

Resumen: *Las aves acuáticas que habitan en colonias durante el período de cría, son particularmente susceptibles a las perturbaciones humanas por sus hábitos conducentes a una alta densidad de nidos. Los factores que disminuyen el éxito reproductivo, incluyen la mortalidad del huevo y el pichón, la evacuación del nido, la reducción de la masa corporal del pichón o crecimeiento lento, el abandonó prematuro del nido por parte de los pichones y comportamientos adultos modificados. Quince especies de colonias de aves acuáticas que nidificaron en 17 colonias del norte y centro de Florida, fueron expuestas a 3 mecanismos de perturbación humana diferentes, a los efectos de determinar distancias de alejamiento recomendables para proteger las agregaciones mixtas de éstas especies. Variaciones intra-específicas e inter-específicas en las distancias de respuesta frente a los mismos mecanismos de perturbación humana. En general, las colonias de aves acuáticas exhibieron una mayor distancia promedio antes de volar en reacción a la cercanía de pasos que al acercamiento de una embarcación a motor. La distancia recomendada de alejamiento fue estimada utilizando una fórmula basada en la media más 1.6495 desviaciones standard de la distancias antes de volar observada, más 40 m* [$\hat{RS} = \exp(\hat{\mu} + 1.6495\hat{\sigma} + 40)$]. *En general, una distancia de alejamiento de alrededor de 100 m para las colonias de aves zancudas y 180 m para las colonias mixtas ("tern/skimmer"), sería adecuada para amortiguar a los sitios que estudiamos de los impactos de las perturbaciones humanas causadas por la aproximación de caminantes y embarcaciones con motor. Recomendamos estudios de seguimiento para probar nuestro modelo en otras colonias de cría.*

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Introduction

Human disturbance can adversely affect wildlife, with colonial breeding birds being particularly susceptible because of their high-density nesting habits. Several studies have shown both qualitative (Johnson & Sloan 1976; Ellison & Cleary 1978; Anderson & Keith 1980) and quantitative (Jenni 1969; Tremblay & Ellison 1979; Parsons & Burger 1982; Kaiser & Fritzell 1984) human impacts on colonial waterbirds. Adverse effects include egg and nestling mortality (Teal 1965; Schreiber 1979; Jeffrey 1987), premature fledging or nest evacuation (Veen 1977), and reduced body mass or slower growth of nestlings (Kurry & Gochfeld 1975; Pierce & Simons 1986). Adult behavior also may be altered by disturbance, resulting in altered foraging patterns (Skagen et al. 1991) and other detrimental effects on reproduction (Gillet et al. 1975; Tremblay & Ellison 1979; Cairns 1980; Safina & Burger 1983). Responses of colonial waterbirds to disturbance may vary with habitat type, physiography of the colony, food supply, seasonality, and bird species (Manuwal 1978; Ollason & Dunnet 1980; Erwin 1989). Some researchers also have reported no significant effects in relation to frequency of disturbance on breeding success (Goering & Cherry 1971) or various degrees of habituation to disturbances (Robert & Ralph 1975; Burger 1981*a*; Burger & Gochfeld 1981).

The increasing popularity of outdoor recreational activities in recent years has resulted in increased human disturbances of colonial waterbird breeding sites. This frequently has led to increased protection of these sites by placing them in public ownership and concomitantly, restricted recreational access to land and water around these breeding colonies. Conservation personnel are faced with the difficult task of effectively buffering important wildlife resources from disruptive human activities despite increasing demands for access to public lands. We define a set-back distance as a minimum distance of nonintrusion by humans measured from the perimeter of a colony that will preclude disturbances to nesting birds. Previous recommendations for set-backs around nesting colonial waterbirds as a strategy to alleviate disturbances have ranged from 50 to 200 meters for tern (Sterninae) species (Buckley & Buckley 1976; Erwin 1989) and 100 to 250 meters for wading bird (Ardeidae) species (Vos et al. 1985; Erwin 1989). Anderson (1988) suggested a "threshold" estimate of 600 meters to protect a Brown Pelican (*Pelecanus occidentalis*) colony in Mexico. Although Florida began protecting colonial waterbird nesting sites from human disturbance in 1976, the set-back distances currently used by a natural-resource personnel to protect avian colonies in Florida are not based on regional empirical data. Most set-back distances were derived from "best estimates" at the time of posting but were difficult to defend on legal and biological grounds. Therefore, a multispecies study was conducted to determine the set-back distances necessary to protect nesting colonial wading birds and seabirds from human disturbance.

We were interested in determining if individual species members of the ground nesting guild (order Charadriiformes) and tree nesting guild (orders Pelecaniformes and Ciconiiformes) would exhibit similar flush distances to the same type of human disturbance. In Florida, the ground-nesting guild is represented by the gulls, terns, and skimmers (family Laridae), and the tree nesting guild is represented by wading birds (families Ardeidae, Threskiornithidae, and Ciconiidae) and other seabirds (families Pelecanidae, Anhingidae, Phalacrocoracidae). One proposed advantage of coloniality is its antipredator role (Lack 1968), and a primary advantage of increased vigilance is to allow the alerted birds to flee (Krebs 1978). Thus, colonial nesting may facilitate group vigilance, and similar species within each colonial nesting guild may exhibit similar flush distances. In addition, we wanted to identify the potential of various types of human activities for causing wildlife disturbance. Pedestrian and boat traffic are the most frequent forms of human disturbance to waterbird nesting in Florida, especially at colonies on islands and marine coastal sites.

Our primary goal in this study, however, was to recommend set-back distances to prevent human disturbance of single-species and mixed-species colonies. Herein, we develop a technique to calculate set-back distances around breeding-bird colonies and recommend this method as a general model that may be applied elsewhere for buffer zones specifically designed for each species and location.

Study Area

Data were collected at eight wading bird and nine seabird (collectively termed colonial waterbirds) nesting sites in Florida during the spring-summer of 1989–1991: Dee Dot Ranch (Duval Co.), a mixed-species wading-bird colony in a sparsely treed freshwater swamp, dominant nesting vegetation was cypress (*Taxodium distichum*); Matanzas Point (St. Johns Co.), two separate tern nesting sites in natural coastal dune habitat, sparsely vegetated with sea oats (*Uniola paniculata*); Port Orange (Volusia Co.), a mixed-species wading-bird colony on a marine dredged-material island, dominant vegetation was black mangrove (*Avicennia germinans*); Oaks Mall (Alachua Co.), a mixed-species wading-bird colony in a freshwater swamp, dominant vegetation was southern willow (*Salix caroliniana*); Lake Yale (Lake Co.), a mixed-species wading-bird colony in a freshwater swamp, dominant vegetation was cypress; Haulover Canal (Brevard Co.), a mixed-species wadingcies of colonial waterbirds to a walking approach that precluded further pooling of the data for the tree-nesting guild into taxonomic groups, such as family and order, higher than the species level. Mean upflight distances for Least Terns (*Sterna antillarum*, 59.0 ± 23.6 m) and Black Skimmers (*Rynchops niger*, 56.2 ± 30.6 m) to a walking approach were about twice the mean individual bird-flush distances (27.9 ± 9.4 m and 25.0 ± 9.6 m, respectively) from their nests after returning from an upflight (Fig. 1).

We found significant ($p < 0.05$) negative relationships—flush distances shorter with successive on-foot approaches—only for Laughing Gulls (*Larus atricilla*) at the Island DS–3D colony ($r^2 = 0.14, p < 0.04, n = 31$) and Cattle Egrets (*Bubulcus ibis*) at the Port Orange colony ($r^2 = 0.66, p < 0.001, n = 12$). We found a significant ($p < 0.05$) positive relationship—flush distances increased with successive on-foot approaches—for individual Black Skimmers at the St. George Island causeway colony during the incubation period of 1989 and 1990 ($r^2 = 0.66, p < 0.001, n = 22$; $r^2 = 0.25, p < 0.001, n = 56$; respectively), but not during the later nestling period. We also investigated autocorrelation (especially first order) using autocorrelation plots, plots of lagged-regression residuals, and Durbin-Watson tests (D) for independence of serial correlation among residuals. We detected significant ($p < 0.05$) positive first-order autocorrelation only for flush distances of individual Least Terns during the incubation period at St. George Island causeway (1989: $D = 0.906$, autocorrelation $= +0.513$, $n = 18$; 1990: $D = 1.543$, autocorrelation $= +0.224$, $n = 77$). Consequently, the individual flush distances for Least Terns and Black Skimmers at the St. George Island causeway colony represented in Figure 1 are the data collected during the nestling period that did not exhibit a significant relationship or autocorrelation between successive walking approaches. As the recommended set-back distances for these two species were calculated from the larger upflight distances (see Fig. 1 and Table 1), these effects are discussed here but did not confound our set-back recommendations.

Boat Approach

Brown Pelicans exhibited the shortest individual flush distance (4 m) and mean flush distance (9.4 ± 5.5 m), whereas Great Egrets (*Casmerodius albus*) possessed the longest mean flush distance (28.9 ± 8.6 m) at the approach of a boat (Fig. 2). As with the walking approach, there were significant differences (ANOVA/Tukey's M.R.T., $p < 0.05$) in the flush distances among some species of the tree-nesting guild that prevented pooling these data into taxa higher than the species level.

Table 1. Recommended set-back (RS) distances between breeding colonial waterbirds and a walking or motor boat approach directly toward the nest.

Order and Species	RS Distance (m)[a] Walking	Motor Boat
Pelecaniformes		
Brown Pelican (*Pelecanus occidentalis*)	76	65
Double-crested Cormorant (*Phalacrocorax auritus*)	96	71
Anhinga (*Anhinga anhinga*)		89
Ciconiiformes		
Great Blue Heron (*Ardea herodias*)	100	82
Great Egret (*Casmerodius albus*)	91	87
Snowy Egret (*Egretta thula*)		67
Tricolored Heron (*E. tricolor*)	88	59
Little Blue Heron (*E. caerulea*)		71
Cattle Egret (*Bubulcus ibis*)		70
Black-crowned Night-Heron (*Nycticorax nycticorax*)	97	
White Ibis (*Eudocimus albus*)	76	
Wood Stork (*Mycteria americana*)	65	63
Charadriiformes		
Least Tern[b] (*Sterna antillarum*)	154	
Black Skimmer[b] (*Rynchops niger*)	178	

[a] *RS distance was calculated by using the formula $\hat{RS} = \exp(\hat{u} + 1.6495\,\hat{\sigma}) + 40$ m. Values were rounded to nearest whole number.*
[b] *RS distances for these species were based on the upflight response.*

Canoe Approach

Double-crested Cormorants (*Phalacrocorax auritus*) flushed at significantly greater distances ($t = 2.580, p < 0.05$) than Anhingas (*Anhinga anhinga*) in our limited data set (Fig. 2).

Comparisons of HDMs

Brown Pelicans ($t = 2.333$), Double-crested Cormorants ($t = 6.168$), Great Blue Herons ($t = 5.172$), and Tricolored Herons (*Egretta tricolor*, $t = 4.351$) exhibited significantly ($p < 0.05$) shorter flush distances to an approaching boat than to walking humans, whereas the Great Egret showed similar flush distances ($t = 1.174, p > 0.05$) to approaches on foot and in a boat (Figs. 1 and 2). Flush distances for canoe and motor boat approaches (Fig. 2) were similar ($t = 1.370, p > 0.05$) for the Anhinga.

Double-crested Cormorants exhibited significantly ($p < 0.05$) greater flush distances than Brown Pelicans for motor boat (t = 8.291) but not for walking (t = 1.140). Great Blue Herons and Great Egrets appear to be two of the more sensitive species when approached on foot or in a boat (Figs. 1 and 2).

Nesting and Interyear Comparisons

There were no significant ($p > 0.05$) differences in flush distances between incubating and brooding adult Brown Pelicans (17.2 ± 10.1 m [n = 38] and 22.5 ± 9.3 m [n = 25], respectively; t = 1.527) and Double-crested Cormorants (24.2 ± 12.6 m [n = 24] and 33.2 ± 14.5 m [n = 14], respectively; t = 1.185) for the walking HDM. There were no significant ($p > 0.05$) differences in flush distances collected between 1989 and 1990 among adult Brown Pelicans for the walking HDM (18.0 ± 6.3 m [n = 45] and 23.0 ± 5.6 m [n = 18], respectively; t = 1.383) and individual flush distances of adult Least Terns (33.4 ± 9.9 [n = 18] and 26.3 ± 10.7 m [n = 53], respectively; t = 1.318), and individual Black Skimmers for the walking HDM (21.5 ± 8.3 m [n = 22] and 28.5 ± 13.6 m [n = 25], respectively; t = 1.188) approaches.

Adult Least Terns (27.9 ± 9.4 m, n = 71) and Black Skimmers (25.0 ± 9.6 m, n = 47) exhibited significantly greater individual flush distances (t = 1.668 and 1.854, respectively; $p < 0.05$) than unattended, mobile nestling terns (17.2 ± 12.3 m, n = 21) and skimmers (16.3 ± 4.4 m, n = 19) during a walking approach. Adult (19.2 ± 7.4 m, n = 63) and nestling (21.3 ± 3.5 m, n = 12) Brown Pelicans exhibited similar (t = 1.523, $p > 0.05$) flush distances to walking approaches.

Discussion

Our study detected interspecific response variation to the same HDM among colonial waterbirds, especially among the tree-nesting guild. Great Blue Herons and Great Egrets generally exhibited the largest flush distances, whereas Brown Pelicans and Wood Storks generally possessed the smallest flush distances. Other studies also have found similar variation among species (Manuwal 1978; Ollason & Dunnet 1980; Burger & Gochfeld 1981; Erwin 1989). Mueller and Glass (1988) noted that some species of waterbirds—Snowy Egrets (*E. thula*), Tricolored Herons, and White-faced Ibises (*Plegadis chihi*)—were more adversely affected by disturbance than other species. We also found that some species, such as Brown Pelicans and Cattle Egrets, were relatively tolerant of human disturbance. This may be due to their long association with and habituation to human activities in Florida; pelicans frequently associate with fishing activities and "panhandle" at docks and piers, and Cattle Egrets often follow farm machinery.

Several species (Brown Pelican, Double-crested Cormorant, Great Blue Heron, Tricolored Heron) exhibited shorter mean flush distances to a boat approach compared to a walking approach. Vos et al. (1985) reported that most boating activity caused the least disturbance to Great Blue Herons. Grubb and King (1991) also found that pedestrian traffic was the human activity most disturbing to Bald Eagles (*Haliaeetus leucocephalus*), and Klein (1993) found that human traffic was more disruptive than vehicular traffic to several species of foraging waterbirds.

We did not detect differences in the flush distances between incubating and brooding adults. Likewise, Erwin (1989) reported no significant correlations between response distances and nesting phase, though he did note a weak (p = 0.10) relationship for Least Terns and nesting phase. In our study, three species (Brown Pelicans, Least Terns, and Black Skimmers) demonstrated that a set-back distance that prevents flushing by nesting adults also should provide an adequate buffer zone to prevent flushing (nest evacuation) by older, mobile juvenile birds. In addition, the upflight distances were greater than the individual flush distances of nesting Least Terns and Black Skimmers and therefore should be used to calculate set-back distances for these species.

We detected both decreasing distance responses to repeated approaches (among Cattle Egrets, Laughing Gulls) and increasing distance responses to sequential approaches (among Black Skimmers), as well as, first-order autocorrelated responses (among Least Terns) during our analyses. It appears that the responses of individuals of these four species to a sequential on-foot approach may have been affected by our previous approaches. Both acclimation to disturbance and increased sensitivity to disturbance phenomena should be considered by researchers in future studies. We could have adjusted for the effects of first-order autocorrelation by multiplying the estimated variance by a function of the estimated autocorrelation coefficient. Because of the much greater distances for the upflight responses of Least Terns and Black Skimmers, however, we did not use data on the flush distances of individual birds for these two species when estimating the recommended set-back distances as we did for the other colonial waterbirds. In addition, we did not estimate set-back distances for Cattle Egrets and Laughing Gulls because of a significant negative relationship in successive approach and flush distances for these two species. To compensate for the effects of acclimation, these species could be represented by a statistic related to the predicted regression value of the first observation, rather than a statistic based on the species mean. However, we did not have enough data to do this. Future research by other investigators should consider these factors when designing and testing hypotheses.

One proposed advantage of coloniality for single-species and multi-species assemblages or nesting guilds is antipredator defense via early warning to colony members (see reviews in Burger 1981*b*; Wittenberger & Hunt 1985). Least Terns and Black Skimmers exhibited similar long upflight distances, an advantage for these ground-nesting guild species. The tree-nesting guild species (Pelecaniformes and Ciconiiformes) showed smaller flush distances, however, possibly the result of nesting above ground level and security from approach of some mammalian predators. These tree-nesting species also exhibited greater interspecific differences in their flush distances. A possible advantage of these mixed-species nesting assemblages would be group vigilance that allows the alerted birds to flee from a predator (Krebs 1978). The intermediate-sized day herons (such as Little Blue Heron, Tricolored Heron, Snowy Egret) that tend to nest under the vegetative canopy (see Burger 1978) would gain an advantage by nesting with the more vigilant (larger flush distances) Great Blue Herons and Great Egrets that tend to nest higher in the canopy.

It appears that acclimation to tangential vehicle traffic also occurs among some colonial waterbirds. Although we made no direct test in this study to determine the minimum distance to elicit a flush response to a tangentially moving vehicle, data from another study demonstrated that Least Terns ($\bar{x}$ = 11.0 m, range = 7–15 m, n = 103 nests) and Black Skimmers ($\bar{x}$ = 12.6 m, range = 9–17 m, n = 61 nests) nested closer (t = 2.247 and 3.225, respectively; $p < 0.001$) to the road edge compared to their individual flush distances (least tern: $\bar{x}$ = 26.6 m, range = 12–59 m, n = 54 nests; Black Skimmer: $\bar{x}$ = 25.0 m, range = 12–44 m, n = 47 nests) in response to walking approaches at the St. George Island causeway colony. The factor limiting nest site proximity to the roadway at this colony was apparently the lack of suitable substrate within the adjacent grass-covered right-of-way. Only rarely did terns and skimmers nesting at the St. George Island causeway colony flush in mass due to nearby (about 10–15 meters) tangential vehicular traffic such as large, noisy tractor-trailers. This suggests that habituation to some types of human disturbance is possible for some species at some sites, especially when breeding habitat is limited as for larids in Florida. Similar instances of acclimation by colonial waterbirds have been reported by Grubb (1978) and Anderson (1988).

Recommendations and Implementation

A major conclusion of our study is that all species must be considered when recommending set-back distances around mixed-species waterbird colonies. Association with mixed-species aggregations may even increase the flushing distances for some species (Stinson 1988). Several authors have recommended set-back distances to protect colonial waterbirds from human disturbance. Vos et al. (1985) recommended buffer zones of 250 meters on land and 150 meters on water for Great Blue Herons. Anderson (1988) proposed a minimum of 600 meters for Brown Pelicans nesting on an island off the west coast of Mexico, but this "minimum threshold" value was derived from indirect estimates of human traffic along footpaths. Schreiber and Schreiber (1978) recommended that Double-crested Cormorants nesting in single-species or mixed-species colonies should not be approached "closer than 75 m." The most extensive study that specifically examined disturbances to colonial waterbirds was by Erwin (1989). Based on a mean (±SD) flushing distance formula, Erwin (1989) recommended a buffer zone of 100 meters for Least Terns and wading birds and 200 meters for Black Skimmers and Common Terns (*S. hirundo*). For terns and skimmers, Erwin (1989) used the upflight response to recommend set-back distances.

Our attempts to quantify the onset of alert and agonistic behaviors by colonial waterbirds in response to exposure to various HDMs proved very difficult. Because of much concurrent nesting activity, it was not always possible to detect when the bird under observation exhibited an alert/agonistic response to the HDM. However, observations from blinds indicated that nesting birds generally became agitated by an approaching disturbance 25–40 meters (n = 11) prior to flushing from the nest. This distance is similar to the additional buffer-zone distance of 50 meters recommended by Vos et al. (1985). The addition of 40 meters (+40 m) to the flushing distances of our sampled populations would be a conservative approach to minimize alert/agonistic responses and likewise allow for effects related to variation in vegetative cover, intraseasonal differences, and food supply that might cause increased stress on the colony (Hunt 1972; van der Zande & Vestral 1985), and other environmental variables that can influence flush distances.

We estimated recommended set-back (RS) distances for individual species of breeding colonial waterbirds calculated from the mean and standard deviation of our sampled populations (Table 1). For a given species, let X_i represent the observed flushing distance for an individual nest approach i and $Y_i = \ln(X_i)$. We assumed that the X_i are independent, identically distributed and follow a lognormal distribution with the parameters μ and σ such that $\mu = E(Y_i)$ and $\sigma^2 = \mathrm{var}(Y_i)$. Using $Q_{0.95}$ as the ninety-fifth percentile of this distribution ($0.95 = P(X_i \leq Q_{0.95})$), the desired RS distance was considered to be $Q_{0.95} + 40$. To estimate $Q_{0.95}$ and the RS, the relationship between percentiles of the lognormal and normal distributions was used. Thus, for the ninety-fifth percentile of a standard normal distribution, $Z_{0.95} = 1.6495$ and

$$Q_{0.95} = \exp(\mu + 1.6495\,\sigma).$$

Then, the estimated $\widehat{RS}$ distance was calculated as

$$\widehat{RS} = \exp(\hat{\mu} + 1.6495\hat{\sigma}) + 40,$$

where $\hat{\mu}$ and $\hat{\sigma}$ are the sample mean and standard deviation for the observed values of $Y_i = \ln(X_i)$, $i = 1, \ldots, n$. We believe that the one-tailed 5% criterion provides a sufficiently conservative margin in the establishment of RS distances for colonial waterbirds while providing a procedure that does not require assumptions that are unreasonable for our data.

Human disturbance during wildlife viewing can subtly disrupt community dynamics (Skagen et al. 1991). Therefore, at mixed-species colonies of waterbirds, the most sensitive species—the most "skittish" species with the greatest flush distance—should be used for determining the RS distance. We further recommend that the upflight distances be used to calculate the RS distance for mixed tern and skimmer colonies. Due to low sample sizes, we were unable to estimate a RS distance for most species disturbed by approach of a canoe. For Anhingas, however, the RS distance for a canoe approach (88 m) is similar to that of a motor boat (89 m). Thus, we tentatively recommend that a RS distance for canoes and other similar vessels be the same as for a motor boat (Table 1). For mixed-species colonies that are subject to multiple HDMs, our data suggest that a RS distance of about 100 meters for wading-bird colonies and about 180 meters for tern/skimmer colonies should provide an adequate buffer zone around the populations we sampled in Florida.

We urge conservation personnel to use prudence when implementing the RS distances in Table 1 for single-species or mixed-species colonies elsewhere. For example, on remote islands seldom visited by humans, terns and other pelagic ground-nesters may be more sensitive than in our study. At the other extreme, some species may exhibit degrees of acclimation to various disturbances for short periods of time (as with the St. George Island causeway colony). But, we believe acclimation phenomena should neither be used as justification for reducing buffer-zone distances nor for attempting to habituate any species to HDMs after birds have colonized a site. Some mitigation may be possible for shorter RS distances when physical barriers prevent direct visual contact between breeding birds and HDMs with low noise levels. Also, some evidence suggests that tangential approach by a HDM (such as vehicular traffic) may allow for a shorter RS distance. This effect may be similar to one observed by Burger and Gochfeld (1981) for Herring Gulls (*L. argentatus*) that responded to the potential threat of approach by a researcher at greater distances if the approach was direct rather than tangential. Henson and Grant (1991) also noted that breeding Trumpeter Swans (*Cygnus buccinator*) only reacted to common vehicular traffic when the vehicles stopped along roadways or sounded their horns. We have observed similar behavioral responses by Least Terns and Black Skimmers at the St. George Island causeway colony and by Cattle Egrets at an I-75 colony during this study.

Conservation personnel must monitor breeding colonies for changes in species composition so that the RS distance can be adjusted for the presence of new, more sensitive species with greater flush distances. Likewise, the perimeter of breeding colonial waterbirds must be monitored annually so that the RS distance reflects current colony boundaries (see Buckley & Buckley 1972; Kerns & Howe 1967; Beaver et al. 1980). Managers also must know if a breeding colony is used as a winter roost to determine if the RS distance should be maintained during the nonbreeding season. If the RS distance is discontinued during the nonbreeding season, it should be re-established several weeks prior to the arrival of breeding birds based on previous monitoring of the breeding chronology of the colony. Effects on the prelaying portion of the breeding cycle associated with disturbance may include disrupted occupation of colony sites (Conover & Miller 1978), subcolony prelaying abandonment (Safina & Burger 1983), or other adverse effects on pair-bond establishment and nest-site selection behavior.

We recommend additional research to examine the effects of variable approach speeds (especially rapid, erratic movements), tangential approaches, presence of seasonal variation in response to disturbance, and other types of HDMs (such as jet-ski vehicles, aircraft overflights, etc.). We realize that there are limits to our method of calculating estimated RS distances for each species and that the values are more subjective than implied from the RS equation. Because of the variation in flush distances among individual birds and species, RS distances may need to be developed on an individual-colony basis. However, we believe the principles and techniques developed here may be applied elsewhere to serve as a general model for specific design of RS distances for each species, location, and situation.

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SHIPSHEWANA LAKE
LaGrange County
Fish Management Report
2002

Introduction

Shipshewana Lake is a 202 acre natural lake located one mile west of Shipshewana, Indiana in LaGrange County. The lake has a maximum depth of 20 feet and an average depth of eight feet. The main inlet to Shipshewana Lake, Cotton Lake Ditch, enters the lake on the north shore and originates in the Cotton Lake watershed. The outlet, Page Ditch, is located on the eastshore of the lake and flows into Taylor Lake which eventually enters Pigeon River. Approximately 50% of the shoreline is developed residentially. A state owned public access site with a concrete boat ramp is located on the south shore of the lake off of County Road 900 West. LaGrange County has a park located on the east shore of Shipshewana Lake. This park provides parking and access for ice anglers at the lake.

An initial general fish population survey of Shipshewana Lake was conducted by Indiana Department of Natural Resources' (IDNR) fisheries biologists in 1968. This survey found a marginal fishery dominated by panfish. No fish management was recommended

Shipshewana Lake was surveyed again in 1975 to evaluate the effects of a large panfish die-off that occurred in 1972. This die off was caused by a bacterial disease. The results of this survey showed a satisfactory fishery was present dominated by bluegill, and black crappie. Once again, no fish management was recommended.

A third fisheries survey of Shipshewana Lake was conducted in 1983. The survey was dominated by golden shiner, black crappie, bluegill and yellow perch. The fishery was again considered satisfactory although increases in the golden shiner and carp populations were noticed along with a decline in water quality. Following this survey it was recommended that northern pike be stocked in Shipshewana Lake to utilize increasing golden shiner and white sucker populations. Pike were subsequently stocked into the lake in 1985 and 1987. A follow-up survey in 1986 showed survival and growth of the 1985 pike was good.

Following the 1983 survey, concerns regarding the water quality resulted in a request filed with the Indiana Department of Environmental Management to investigate The Shipshewana Lake watershed. The purpose of this investigation was to discover possible sources of nutrients entering the lake and propose control methods..

As a result, International Science and Technology was contracted to conduct a feasibility study regarding the restoration of Shipshewana Lake. This study, which was funded by the Shipshewana Community Lake Improvement Association and the IDNR's Lake Enhancement Program, was submitted in 1989. This report identified sources of nutrients contributing to water quality declines at Shipshewana Lake and potential restoration measures. A dredging project was recommended at Shipshewana Lake to remove the large amount of sediment that covered the lake bottom and continually recycled unwanted nutrients throughout the system. This dredging project was finally carried out in 1999. Using a suction dredge, the contractor was able to remove approximate 227,500 cubic yards of sediment from the lake bottom. While the southern basin of the lake was considered free of sediment, it was estimated up to five feet still remained in the north basin due to lack of funds to finish the project. In addition to the dredging project, farmers in the watershed were taught and encouraged to use agricultural conservation practices. The total cost of the project, which was funded by Build Indiana Fund monies channeled through the Division of Soil Conservation, was approximately $2.4 million.

An additional fish population survey was conducted at Shipshewana Lake in 1989 to check the status of the fishery. During this survey a total of 2,504 fish representing 14 species were collected. The dominant species by number was golden shiner (40.6%) followed by black crappie (29%), bluegill (12.3%) and largemouth bass (6.5%). By weight, carp dominated (34.1%) followed by largemouth bass (21%), spotted gar (10.5%), golden shiner (8.2%) and black crappie (8.2%). In addition. 24 northern pike were collected. Twenty-one of these fish were age II+ fish from the 1987 stocking. The largest pike collected measured 33 inches in length. It was determined that although a good sport fishery still existed in Shipshewana Lake, there was a decline in the quality of the bluegill and black crappie populations. The golden shiner population had also exploded. The largemouth bass population appeared very healthy. Five year classes were collected and growth rates were above average. In addition, over 41% of the bass sample was comprised of fish 14 inches in length or larger, which is considered harvestable size. This is outstanding for northern Indiana natural lakes.

The current survey was conducted from June 3-7, 2002. Fish collection methods included gill netting, trap netting and nighttime D.C. electrofishing.

Results

A total of 1,424 fish representing 15 species were collected during the survey. Black crappie was the dominant species by number (34.2%) followed by bluegill(28.3%), golden shiner (8.4%), yellow perch (7.4%) and largemouth bass (7.3%). Carp dominated the sample by weight (36.2%) followed by white sucker (12.21%), black crappie (12.16%), spotted gar (10.5%), bluegill (8.6%) and largemouth bass (8.2%). The total weight of fish collected was 917.22 pounds.

Black crappie ranked first among all species collected in number (34.2%) and fourth by weight (12.16%). They ranged in length from 2.7 (age I+) to 10.2 (age V+) inches and averaged 7.7. The 487 crappie collected weighed 111.49 pounds black crappie 8.5 inches in length or larger, considered harvestable size, comprised 38.6% of the sample. Growth rates for all ages of crappie were average. In 1989, crappie were second in abundance behind golden shiners but were first among game species collected. Only 10.2% of the 725 crappie collected that year were harvestable size. Crappie were also second in abundance in 1983, again following golden shiners. Harvestable size crappie comprised only 7.1% of the sample in 1983.

A total of 403 bluegill weighing 79.06 pounds were collected during the survey. They ranged in length from 3.2 (age I+) to 13.0 (age IV+) inches and weighed 8.57 pounds. Bluegill comprised 28.3% of the sample numerically, which ranked second, and 8.6% by weight, which ranked fifth. Approximately 39% of the bluegill collected were harvestable size, 6 inches in length or larger.

Growth rates for bluegill were average. In 1989 bluegill were third numerically and only 5.2% of these were harvestable size. Bluegill were also third in abundance in 1983, however 80.6% of these fish were harvestable size.

Yellow perch ranked fourth in abundance (7.4%) among species collected. The 106 perch captured weighed a total of 33.13 pounds. They ranged in length from 6.9 (age II+) to 11.8 (age V+) inches and averaged 8.4. Harvestable size perch (eight inches in length or larger) comprised 62.3% of the sample. Perch were fourth in abundance (9.2%) in 1983 but by 1989 the population had declined to the point where only five were collected. Although 153 perch were collected in 1983, only 5.9% of these were harvestable size.

A total of 104 largemouth bass weighing 74.89 pounds were collected during the survey. Bass ranked fifth by number (7.3%) and sixth by weight (8.2%) among species collected. They ranged in length from 4.5 (age I+) to 19.1 (age VII+) inches and averaged 10.4. Bass 14 inches in length or larger, considered harvestable size, comprised 9.6% of the bass sample. Growth rates for all bass year classes except age IV+ fish was above average for northern Indiana natural Lakes. The age IV+ fish exhibited average growth. Bass ranked fourth by number in 1989 (6.5%) and seventh in 1983 (3.5%). Almost three times as many bass (163) were collected in 1989 as in 1983 (58). However, the percentage of harvestable size bass was similar for each year at 37% in 1983 and 41% in 1989. Both of these percentages are very high for northern Indiana natural lakes. Bass growth rates for both these years were well above average.

One other game species was collected during this survey, one channel catfish measuring 20.3 inches in length and weighing 3.16 pounds. This marks the first time a channel catfish has been collected from Shipshewana Lake.

Summary

It appears the lake restoration project at Shipshewana Lake has had a positive impact on the fishery. The lake currently supports a good sport fish population. Black crappie and bluegill now dominate, as together they comprised 63% of the sample numerically and 21% by weight.

Yellow perch numbers have also increased (7.4%) and largemouth bass numbers remain good.(7.3%). The percentage of harvestable size bass in the population remains outstanding. Growth rates for sport species range from average to above average and harvestable size fish are plentiful. Golden shiners, while still present, have fallen to a level where they now comprise only 8% of the fish population by number, ranking third. This compared to the previous two surveys when they were the dominant species by number, comprising 30% of the sample in 1983 and 41% in 1989. Over 1,000 golden shiners were collected in 1989 while only 119 were collected during the current survey. One negative has been the disappearance of the northern pike. After collecting 24 pike in 1989, none were collected during the current survey. Another negative concerns carp. They once again dominated the sample by weight (36.2%) similar to 1989 when they comprised 34.1% of the sample by weight.

No fish diseases were detected at Shipshewana Lake during the survey but black spot was evident on many of the perch collected. Black spot is caused by a parasite called a fluke that spends part of it's life cycle burrowed into a fish: The fish reacts by producing a black pigment to surround the fluke. The fluke is harmless to humans and cannot be transferred to them through eating infested fish. Shoreline erosion at Shipshewana Lake is minimal.

Recommendations

No fish management is recommended for Shipshewana Lake at this time.

Submitted by: Larry A. Koza, Fisheries Biologist

Date: 4/14/03

LAKE SURVEY REPORT

Type of Survey
[ ] Initial Survey [X] Re-Survey

Lake Name	County	Date of survey (Month, day, year)
Shipshewana Lake	LaGrange	06/03-07/02
Biologist's name		Date of approval (Month, day, year)
Larry A. Koza		

LOCATION

Quadrangle Name	Range	Section
Shipshewana	8E	3, 4, 9, 10
Township Name	Nearest Town	
37N	Shipshewana	

ACCESSIBILITY

State owned public access site	Privately owned public access site	Other access site
Cement boat ramp		

Surface acres	Maximum depth	Average depth	Acre feet	Water level	Extreme fluctuations
202	20 Feet			855.08 MSL	None

Location of benchmark
Southeast shore

INLETS

Name	Location	Origin
Cotton Lake Ditch	South	Cotton Lake
Mud Lake Ditch	West	Mud Lake

OUTLETS

Name	Location
Page Ditch	East through Taylor Lake to the Pigeon River

Water level control

POOL	ELEVATION (Feet MSL)	ACRES
TOP OF DAM		
TOP OF FLOOD CONTROL POOL		
TOP OF CONSERVATION POOL		
TOP OF MINIMUM POOL		
STREAMBED		

Bottom type
- [ ] Bolder
- [X] Gravel
- [X] Sand
- [X] Muck
- [ ] Clay
- [ ] Marl

Watershed use
General farming and residential.

Development of shoreline
Approximately 50% of the shoreline is residentially developed.

Previous surveys and investigations
IDNR Fisheries Surveys: Hudson, 1968; Peterson, 1975; Ledet, 1983; Koza, 1986; Ledet, 1989.